

STUDIES OF THE EFFECTS OF NET FRAGMENT ENTANGLEMENT ON NORTHERN FUR SEALS
PART 2: SWIMMING BEHAVIOR OF ENTANGLED AND NONENTANGLED FUR SEALS

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ABSTRACT

The effects of net fragment entanglement on the swimming behavior of fur seals were observed. Net fragments of six different weights (0.5 to 3 kg) were attached to the necks of eight fur seals, two males and six females, 4 to 9 years old. They were released in an aquarium pool with fish, and their swimming speed and time required to capture a fish were recorded. Of the eight individuals examined, three showed active feeding behavior. As the amount of attached net was increased, swimming speed decreased and more time was required for an entangled seal to catch a fish. Decrease in swimming speed was proportional to the relative load of net fragments (net weight/body weight).

INTRODUCTION

Marine debris is known to cause problems for various animals such as fish (High 1985), marine mammals (Calkins 1985; Henderson 1985), seabirds (Tull et al. 1972), and turtles (Balazs 1985; Cawthorn 1985). Many fur seals have been found on the breeding islands entangled in fishing net fragments and packing bands (Waldichuk 1978; Scordino 1985). Fowler (1982) noticed that entanglement probably was a cause of recent decline in the Pribilof population of the northern fur seal, *Callorhinus ursinus*. In order to understand the mechanism and impact of net entanglement on northern fur seals, the National Research Institute of Far Seas Fisheries has conducted various experiments at the Izu-Mito Sea Paradise, an aquarium, since 1983. The mechanism of entangling and influence of entanglement on activity patterns of fur seals were surveyed before (Yoshida et al. 1985,

1990). In this study, experiments were conducted to understand the effect of net entanglement on swimming behavior of fur seals.

MATERIALS AND METHODS

Swimming speed and feeding behavior of net-entangled fur seals were observed in a pool of the Izu-Mito Sea Paradise in Numazu, Japan, from 27 January to 19 February 1986. The pool was 22 m wide, 10 m long, and 4 m deep (Fig. 1). The front of the pool was made of transparent plexiglass through which underwater movements of fur seals were observed. Trawl net fragments of six different weights were attached to the necks of eight fur seals, two males, and six females, estimated to be from 4 to 9 years old (Table 1). The nets used were gray trawl nets made of polyethylene, with a twine size of 3.4 mm and a mesh size of 24 cm. Specific gravity of the net was 0.77, which meant that a net fragment weighing 1 kg of air had a buoyancy of 340 g in seawater with a specific gravity of 1.03. Weights of the six fragments were 0.5, 1.0, 1.5, 2.0, 2.5, and 3.0 kg. As a control, free-swimming animals were also observed.

Experiments were conducted between 1600 and 1730 every day. Fur seals had not been fed since the previous day so that they would respond readily to food. Measurement of swimming speed was conducted in the following manner. One seal was released into the pool and lured to one corner by a display of food. Then a man showed a fish and threw it 8-10 m from the seal. When the seal started swimming after the fish, one observer recorded the time taken by the seal to swim a distance of 6 m. The distance was measured using the interval of frames supporting the glass wall. Each individual was tested using nets of two different weights per day. For each weight of net, an individual was obliged to swim eight times. One seal could make up to 16 swims in a day. If a seal would not chase a fish, it was removed from the pool and another individual was introduced.

Time to capture a fish was measured for the three seals which readily swam for a fish (M1, M2, and F1 in Table 1). Basic design of the experiment was the same as that for swimming speed measurement. Live sardine, *Sardinops melanostictus*, 12.5 to 15.0 cm in length and 15-29 g in weight, was used as bait and was thrown 8-10 m ahead of the seal. The time it took the seal to catch the fish was measured. Eight trials were made for each net weight, although the number was reduced when an animal with heavy entanglement looked tired.

To evaluate the effect of net entanglement on the basis of body weight, relative load of attached net was calculated:

$$\text{relative load} = \frac{\text{weight of attached net}}{\text{body weight of fur seal}} \times 100 (\%)$$

During the experiment period, average air temperature was 7.1°C (1.1°-11.0°C), average water temperature was 12.7°C (12°-13.4°C), and average humidity was 57.4% (36-83%).

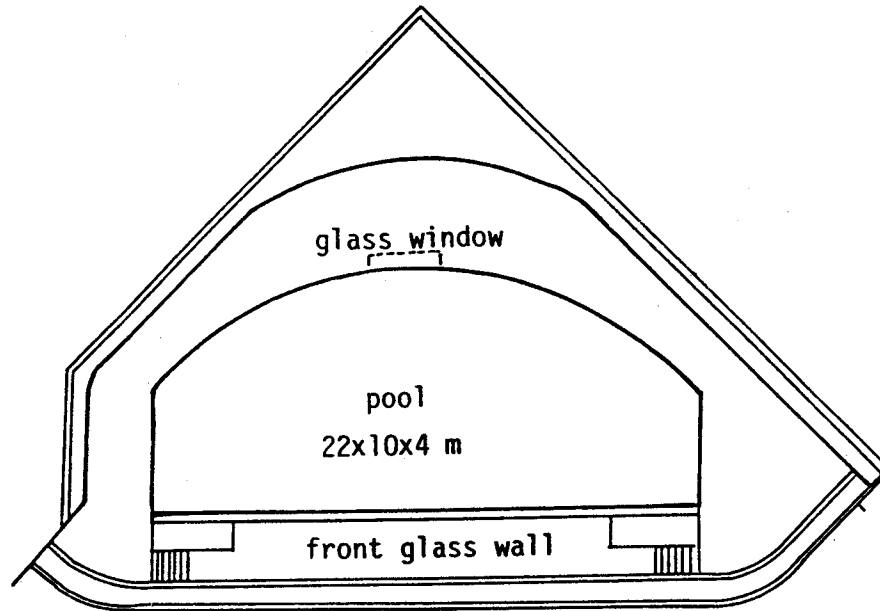


Figure 1.--The pool used in this study. Underwater movements of fur seals were observed through the glass wall.

RESULTS

Individuals Used in the Experiment

Of the six female fur seals used, only one individual (F1) swam actively for fish. She chased fish a total of 32 times for swimming speed measurement, whether she was free-swimming or entangled in 0.5 to 1.5 kg of nets. Two other females (F3 and F5) swam only once or twice, even when they were free from nets. The remaining three made no attempt to swim after a fish. Two males (M1 and M2) tried to catch a fish in every case, whether free-swimming or entangled in up to 3.0-kg nets.

General Behavior

Fur seals often swam on their backs when they were free from entanglement, but entangled seals did not exhibit this type of swimming. All the seals tested were able to swim down to the bottom of the pool (4 m deep) in every degree of entanglement. When they chased a fish, they swam in the upper layer within 1 m of the surface. When entangled seals swam, their bodies twisted up and down. The body undulation was intensified as the amount of attached net was increased.

Swimming Speed

Swimming speed of the three fur seals (M1, M2, and F1), calculated from the amount of time it took them to pass the 6-m mark, decreased as the weight of attached nets increased (Table 2, Fig. 2). The average swimming speed of M1 without entanglement was 2.98 m/sec, but fell to 1.05 m/sec

Table 1.--Information on the fur seals used in the experiment.

Seal	Sex	Capture		Age and size at experiment			
		Date	Location		Age (year)	Body length (cm)	Body weight (kg)
			(Lat.)	(Long.)			
M1	M	July 1981	Robben Island		4	137	58.0
M2	M	4 Mar. 1982	36°34'N	141°14'E	4	132	54.0
F1	F	July 1981	Robben Island		4	120	23.0
F2	F	10 May 1980	37°57'N	142°14'E	6 ^a	120	35.5
F3	F	8 Mar. 1982	36°26'N	141°06'E	6 ^a	121	30.5
F4	F	4 Mar. 1982	36°42'N	141°15'E	9 ^a	125	36.0
F5	F	8 Mar. 1982	36°27'N	141°10'E	8 ^a	124	38.0
F6	F	9 Mar. 1982	36°30'N	141°16'E	7 ^a	122	32.5

^aEstimated age.

when 3-kg nets were attached. That of M2 decreased from 3.04 to 0.96 m/sec when 3-kg nets were loaded. For both M1 and M2, the speed was about one-third of that in a nonentangled state. The average swimming speed of F1 free from entanglement was 2.51 m/sec, but it fell to 0.73 m/sec when 1.5 kg nets were attached.

Figure 3 shows the relation between average swimming speed and relative load of net fragments. The relationship was similar for the three individuals. Swimming speed decreased in proportion to the relative load of attached nets. Linear regression of the relationship between relative load and swimming speed was

$$\text{swimming speed (m/sec)} = 2.26 - 0.25 \times \text{relative load (\%)} \quad (r = -0.97)$$

Swimming speed of free-swimming animals was excluded from the regression.

Time Required to Capture a Fish

Table 3 shows the time it took for three seals, M1, M2, and F1, to capture a fish. F1 was not tested with net fragments heavier than 1.5 kg because that much weight seemed too heavy for her. The relationship between weight of nets and time required to capture a fish is shown in Figure 4. Although there was a considerable range, all three seals required more time to catch a fish as the amount of attached net was increased. For the three weights of nets examined, average capture time was the longest for F1. It was observed that when fur seals tried to catch a live fish, they approached the fish and turned their heads quickly to snap at it. Entangled seals had difficulty with the dash and snap.

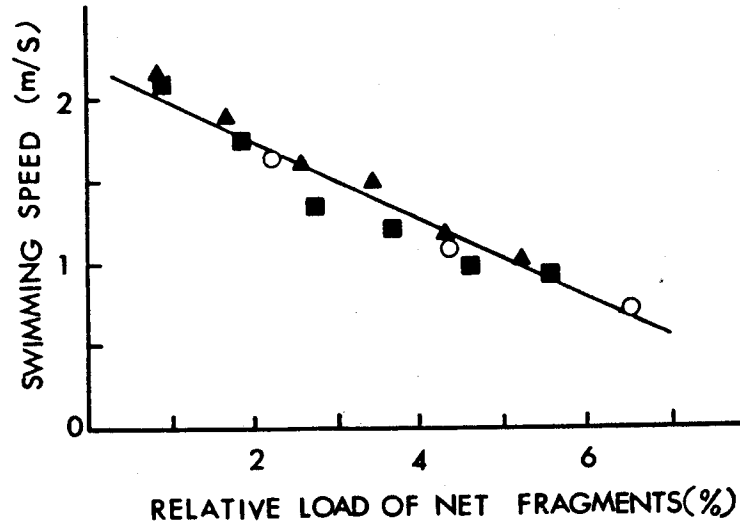


Figure 2.--Changes in swimming speed of three fur seals due to net entanglement. M1:▲ M2:■ F1:○.

Table 2.--Changes in swimming speeds (m/sec) of three fur seals due to amount of net entanglement. Speeds were calculated using the time it took the seals to swim a distance of 6 m.

Seal	Weight of net (kg)						
	0.0	0.5	1.0	1.5	2.0	2.5	3.0
	Swimming speed						
M1							
Mean	2.98	2.16	1.92	1.63	1.50	1.20	1.05
Minimum	2.72	2.06	1.71	1.50	1.39	1.11	0.95
Maximum	3.15	2.40	2.06	1.76	1.66	1.27	1.20
Standard error	0.14	0.11	0.13	0.10	0.08	0.06	0.08
Sample number	8	8	8	8	8	8	8
M2							
Mean	3.04	2.09	1.77	1.37	1.23	1.01	0.96
Minimum	2.60	1.87	1.66	1.25	1.13	0.93	0.82
Maximum	3.33	2.30	1.87	1.57	1.33	1.17	1.09
Standard error	0.29	0.15	0.07	0.10	0.07	0.07	0.11
Sample number	8	8	8	8	8	8	8
F1							
Mean	2.51	1.66	1.09	0.73	--	--	--
Minimum	2.30	1.50	0.90	0.58	--	--	--
Maximum	2.85	1.93	1.25	0.89	--	--	--
Standard error	0.23	0.16	0.15	0.12	--	--	--
Sample number	6	6	6	6	0	0	0

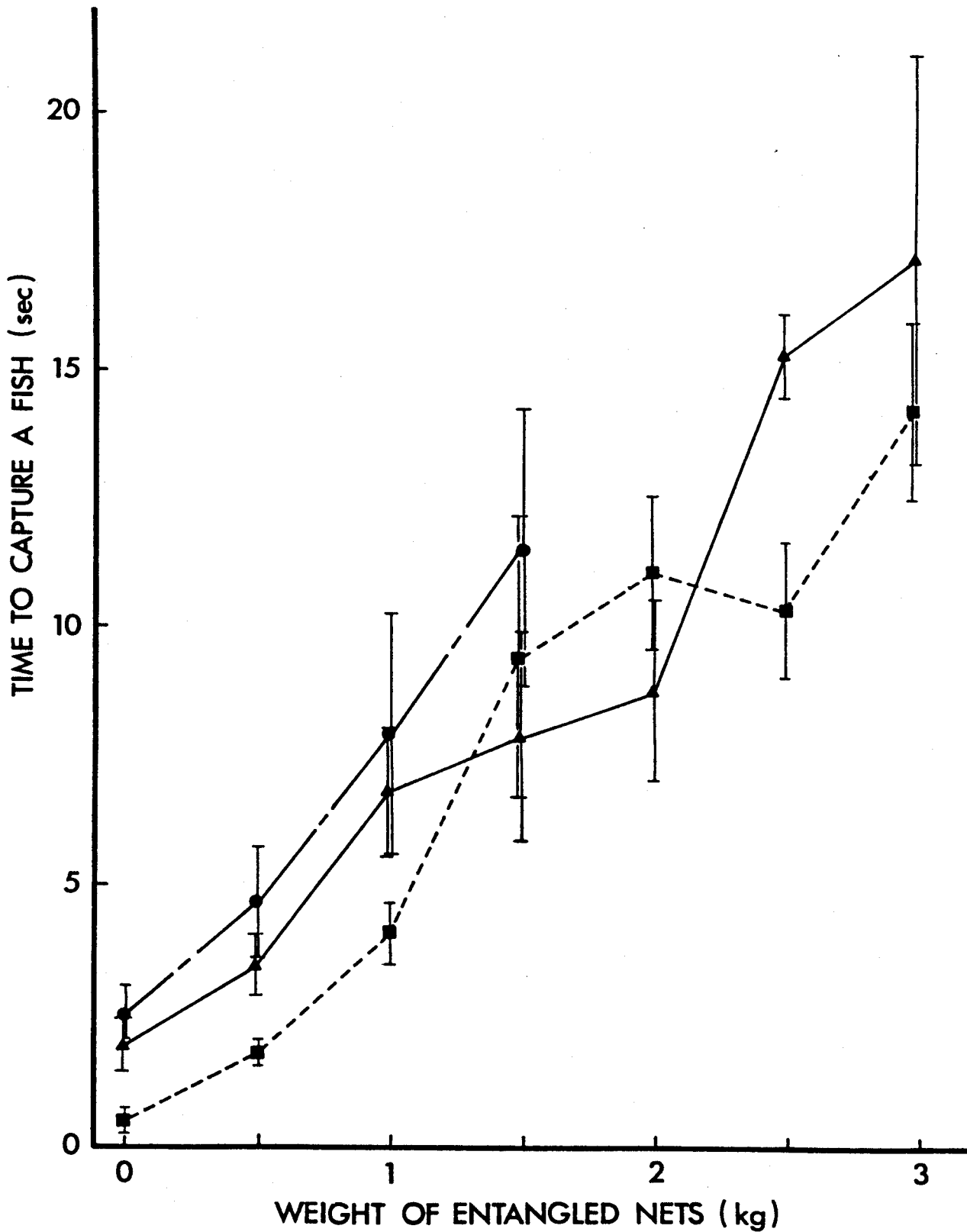


Figure 3.--Relationship between average swimming speed and relative load of attached net. M1: \blacktriangle — \blacktriangle M2: \blacksquare --- \blacksquare F1: \bullet — \bullet

Table 3.--Time to capture a live fish, *Sardinops melanostictus*, required by the three seals carrying different weights of nets.

Seal	Weight of net (kg)						
	0.0	0.5	1.0	1.5	2.0	2.5	3.0
	Capture time (seconds)						
M1							
Mean	18.1	34.1	68.1	78.8	87.8	152.8	171.6
Range	4-47	11-62	9-115	15-200	30-178	135-191	35-346
Standard error	5.2	5.9	12.1	21.0	17.3	8.0	39.4
Sample number	8	8	8	8	8	8	6
M2							
Mean	4.4	17.6	40.6	94.3	110.9	103.6	142.2
Range	3-5	9-30	15-71	12-247	58-180	67-150	76-210
Standard error	0.3	2.5	5.9	27.4	14.9	13.0	17.0
Sample number	8	8	8	8	8	6	6
F1							
Mean	25.1	46.3	79.0	115.3	--	--	--
Range	14-59	16-97	17-218	30-264	--	--	--
Standard error	5.3	10.7	23.7	27.0	--	--	--
Sample number	8	8	8	8	0	0	0

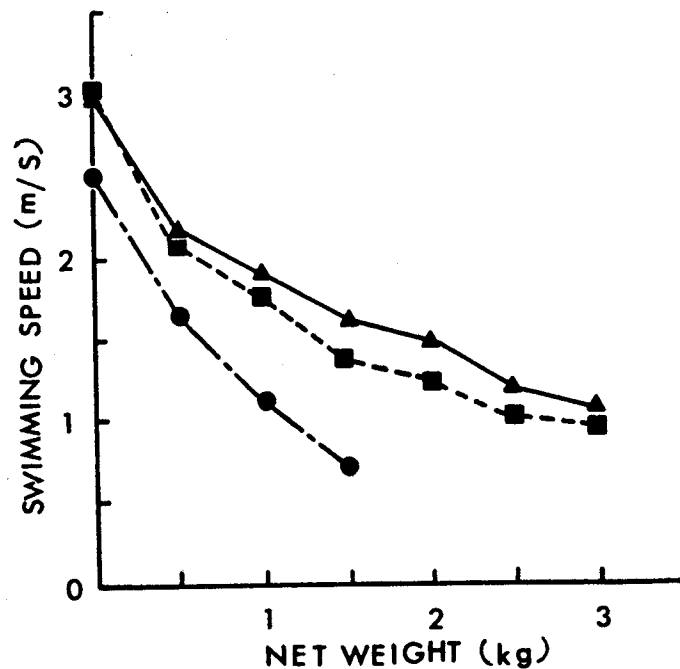


Figure 4.--Relationship between weight of nets and time required to capture a fish. Vertical lines indicate standard error. M1:▲—▲ M2:■---■ F1:●---●

DISCUSSION

Of the eight animals used in these experiments, only three 4-year-olds, one female and two males, tried to capture a thrown fish. M1 and F1 were brought from Robben Island in 1981 as pups and were fed milk by men. All the other seals were caught pelagically. Difference in tractability of the seals might be derived from individual history as well as age, sex, hunger, and disposition.

Swimming speed of the entangled seals decreased as the weight of nets increased. Negative linear relation was observed between swimming speed and relative net load. Decrease in swimming speed might result from two physical forces of net fragments: buoyancy and drag. Buoyancy lifted the body at the neck and shifted the center of gravity. Buoyancy of nets is likely to hinder the dives of entangled seals although all the seals could dive to the bottom of the pool. Body undulation of entangled seals, which was observed when they swam, might be brought about by lifting of the neck caused by buoyancy. At the same time, swimming efficiency was reduced by the drag of the attached nets. These two forces would interfere with diving and swimming and would increase the energy expenditure of entangled seals.

Entangled seals took longer to capture a live fish as the weight of attached nets increased (Fig. 4). The increase in the capture time was derived from a decrease in swimming speed and hindrance of quick body motion. This result indicates that foraging efficiency of entangled seals will be lower than that of free-swimming seals. Heavily entangled seals should suffer from a large expenditure and a small intake of energy. Such an energy problem may be a cause of mortality of entangled seals as well as traumatic damage.

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SIMULATING THE ROLE OF ENTANGLEMENT IN NORTHERN FUR SEAL,
CALLORHINUS URSINUS, POPULATION DYNAMICS

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ABSTRACT

A multiage class model treating populations of both male and female fur seals was developed to examine the plausible long-term effect of their entanglement in discarded net debris. The model is based on the data available on age-specific survival and fecundity including data supporting the assumption of density-dependent survival of pups on land and of juveniles up to age 2 at sea. Also included in the model are age-specific and sex-specific harvests for the subadult male harvest as well as other pelagic and land-based commercial and scientific harvests. Entanglement in the model is linked to the observed incidence of subadult males in the harvest (or roundup). Supporting work in model development and parameter estimation has involved evaluation of various attempts to estimate both juvenile survival at sea and the mortality rate due to entanglement. This evaluation work has considered the appropriateness of assumptions and statistical tests used. Model results were evaluated by comparison with survey estimates of pup abundance and of harem bulls on the Pribilof Islands for years when these were available (between 1912 and 1960). Post-1960 survey results were used to examine the plausibility of entanglement mortality estimates in predicting the observed fur seal abundance decline. Sensitivity analysis on the model is used to indicate areas where there is a need for either further data collection or further analysis of existing data.

INTRODUCTION

The marked decline in northern fur seal, *Callorhinus ursinus*, populations on the Pribilof Islands since the mid-1970's has been attributed to a variety of causes, one of the most compelling of which is increased mortality due to seals' entanglement in discarded fishing net

In R. S. Shomura and M. L. Godfrey (editors), Proceedings of the Second International Conference on Marine Debris, 2-7 April 1989, Honolulu, Hawaii. U.S. Dep. Commer., NOAA Tech. Memo. NMFS, NOAA-TM-NMFS-SWFSC-154. 1990.

debris (Fowler 1985, 1987) and other flotsam (e.g., packing bands). Although an earlier decline (1957-64) was probably due to pelagic scientific sampling of females at sea and harvest of females on land, later continued declines have not been explicable as being linked to repercussions of this harvest (York and Hartley 1981).

From observations of numbers of subadult male seals in the harvest entangled (about 0.4%), Fowler (1982), using a simple differential equation model and assumptions about the length of time entangled seals survive and the ratio of seals tangled in large (>0.4 kg) versus small (<0.4 kg) debris pieces (the seals entangled in large debris being assumed to die before reaching land), gave predictions for the possible effects of entanglement on the population. These predictions indicated that the entangled seals observed in the harvest could account for annual seal mortalities as high as 17%. Swartzman (1984) expanded Fowler's model to include age classes and density-dependent pup survival on land, as reflected in data from Lander (1981). Swartzman (1984) showed that the age and duration of susceptibility to entanglement can affect the annual mortality rate due to entanglement. The worst case scenarios (i.e., 2 months or less for half the entangled seals to die with only ages 1-3 susceptible to entanglement, or less than 12 months for half the entangled seals to die when all age classes are susceptible to entanglement) result in a long-term elimination of the fur seal population.

We have developed a model to investigate the effect of entanglement on fur seal population dynamics. This model separates male and female populations by age class and separates each age-sex class into entangled and unentangled animals. Sex-specific and age-specific susceptibility to mortality and entanglement mortality rates are also considered. Annual entanglement rate in small (<0.4 kg) debris is grounded in the observed fraction of entangled subadult males in the harvest. In the long-term simulation, harvests of males and scientific samples of females are removed from the population as an amount of seals (rather than as a rate).

The population dynamics of fur seals have been the object of many studies, and several models have been built in this regard. The present model is like many in that it is age-structured. Table 1 gives an overview of these previous models. Current modeling work is motivated by the need to synthesize current entanglement information and by the lack of previous treatment of the male population, no inclusion of male harvests and no previous formal sensitivity analysis having been done on previous models. Also, earlier entanglement models are equilibrium models that, while they were based on the best parameter values available at the time they were constructed, were not evaluated by being compared with historical data on pup estimates and bull counts. Finally, our model brings many of the parameter estimates up to date by including the latest available data. The large number of previous models points to the excellent long-term data base on fur seals, although, as will be shown later, assumptions must be made to fill gaps in the data when long-term model projections are made.

Table 1.--Comparison of northern fur seal population dynamics models (F = female, M = male).

Model	Age range/sex	Years	Harvest	Comments
York and Hartley 1981	2-25/F	1956-79	1956-74F	Juvenile survival higher after 1979.
Smith and Polacheck 1981	3-20/F	Not run	--	Challenged differential M/F juvenile survival.
Eberhardt 1981	1-22/F	1952-77	1952-68F	Density-dependent survival to age 3.
Fowler 1982	Pooled	Equilibrium	None	Entanglement effect with several mortalities.
Swartzman 1984	3-20F	Future, equilibrium	None	Density-dependent survival on land.
Trites 1984	1-25/M&F	1950-80	1956-74F constant M	Leslie model sensitivity analysis.
Reed and French 1987	1-29/M&F	1912-2000	1956-76F constant M	Density-dependent pup and juvenile survival.
Swartzman and Huang	2-18M 2-25F	1911-86	1956-76F Subadult male	Entanglement.

MODEL DOCUMENTATION

The model consists of 24 female and 18 male age classes. Populations in each modeled age class are updated by age-specific and sex-specific survival. Sex-specific and year-specific harvests on either (or both) land and sea are also removed from the proper age-sex classes each year. Pup numbers are computed from the adult female population based on age-specific fecundities. Running the model consists of solving a set of differential equations (one for each age-sex class) using the Runge-Kutta method with a time step of 0.25 years. The model is run from 1911 to 1986. The 0.25 time step was chosen to fit with the time period that pups are on land in the Pribilof Islands (3 months), allowing computation of pup survival on land to occur over a single time step. Survival of juveniles from the time they leave land to age 2 is also modeled as a density-dependent factor

based on regression analysis of pup counts and male survival estimates (data provided in Lander 1981). Entanglement rate depends on a year-specific susceptibility (based on the observed proportion of entangled subadult males in the harvest each year), an age-specific relative susceptibility factor, and the ratio of seals entangled in large to seals entangled in small debris parameter. Additionally, there are age-specific and sex-specific entanglement mortality and escapement (from entanglement) rates. Animals entangled in large debris (>0.4 kg) are assumed to die rapidly (at the same rate at which they are entangled).

A mnemonic notation is used to describe model equations (Swartzman and Kaluzny 1987). The first digit of a variable name denotes the variable type, with x used for state variables, k for parameters, g for intermediate variables, and z for driving variables (unaffected by system behavior and read in from a driver data file). The following letters are descriptive mnemonics such as mrt for mortality or n for numbers. Several parameters are numerically subscripted (e.g., k_1) rather than having a mnemonic name. This was done for parameters that were not easily made mnemonic.

The model is a series of differential equations for the rate of change of female and male seals by age class, with entangled seals (in small net debris) separated from unentangled seals.

$$\frac{dxn_{ij}(t)}{dt} = -(gent_i(t) + kmrt_{ij} + gmrtlg_{ij}(t))xn_{ij}(t) + kesc_{ij}xent_{ij}(t) \quad (1)$$

where $j = 1$ for male and 2 for female

i is an age index (1-24 for female; 1-18 for male. These denote ages 2-25 and 2-19 for females and males, respectively)

xn_{ij} - number of unentangled seals in age class i of sex j

$gent_i$ - entanglement rate in small debris for age i seals (yr^{-1})

$kmrt_{ij}$ - natural mortality rate for age i sex j seals (yr^{-1})

$gmrtlg_{ij}$ - entanglement rate in large debris for age i sex j seals (yr^{-1})

$xent_{ij}$ - number of entangled seals in age class i of sex j

$kesc_{ij}$ - rate of escapement from entanglement for age i sex j seals (yr^{-1}).

Population dynamics of entangled seals are:

$$\frac{dxent_{ij}(t)}{dt} = -(kmrtent_{ij} + kmrt_{ij} + gmrtlg_{ij}(t) + kesc_{ij})xent_{ij}(t) + gent_i(t)xn_{ij}(t) \quad (2)$$

where $kmrtent_{ij}$ = entanglement mortality rate for seals entangled in small debris (yr^{-1}).

These equations account for the possibility of entangled seals escaping as reported by Scordino and Fisher (1983), Fowler (1987), and Fowler et al. 1990).

In addition to these continuous time equations there are also discrete time equations that update the population at set times of the year.

- The pupping time (i.e., early July), when pups are produced, ages of the populations are updated, and harvests of subadult males (and pelagic harvests, if any) are taken;
- The time pups leave land, when the density-dependent number of surviving pups is computed.

The number of pups produced in any year are computed from age-specific fecundity

$$xpup(t) = \sum_{i=1}^{24} kfec_i xn_{i2}(t) \quad (3)$$

where $kfec_i$ = age-specific fecundity including the influence of age-specific maturity.

At the time pupping occurs, the model updates time, ages the seals by 1 year, and removes seals by harvest for that year (harvest includes the subadult male commercial harvest, any harvest of females, and any scientific samples taken that year).

$$xn_{ij}(t+1) = xn_{(i-1)j}(t) - zharv_{ij}(t+1) \quad (4)$$

where $zharv_{ij}(t)$ = the total harvest and samples of age-sex class ij in year t (data entry).

Harvests are most commonly applied to the annual subadult males on the Pribilof Islands, but involved females between 1956 and 1968 and research samples including a variety of age-sex classes in many years. Analogous harvest equations exist for entangled male and female animals (there is a data file for entangled seals as well as for unentangled seals from the harvest statistics).

The survival of 1-year-old and 2-year-old seals is treated somewhat differently from the survival of older seals. The natural mortality of seals between age 3 months (the time pups leave land) and age 2 is computed by the density-dependent function

$$gmrt_{mj}(t) = kfmrt_j \frac{-\log_e \left[\frac{ks_1(xpup(t-m+0.25) - ks_2)}{(xpup(t-m+0.25) - ks_3)} \right]}{1.75} \quad (5)$$

Here m denotes age class (1 or 2) and j sex class. The 0.25 year (3 months) adjusts time back to the time pups leave land. The ratio of female to male mortality rates is $kfmrt_2$, and $kfmrt_1 = 1$. The total number of pups leaving land (male + female) in the year for which we are computing mortality rate ($m = 1$) or in the previous year ($m = 2$) is $xpup(t-m+0.25)$. Instead of $kmrt_{mj}$, $gmrt_{mj}(t)$ is used in equations (1) and (2) for age classes 1 and 2 to denote that these are intermediate variables rather than parameters. The natural logarithm and 1.75 are used to convert the fraction of seals surviving to age 2 (excluding entanglement) to a rate. A mortality rate must be used instead of a fraction surviving (which is what we estimate from the primary data source) because entanglement mortality may also be incurred by these younger seals. We noted earlier that the model considers age classes beginning with age 2 seals. As such, the above computations for age class 1 seals are not included in the part of the code that deals with the seal age classes, but as a separate calculation. Age class 1 animals are excluded from the model because very little is known about survival rates of pups after they leave land and estimates are based solely on the male juveniles that begin showing up in the Pribilofs at age 2.

At the time pups leave land (at 3 months of age), the model computes the number of pups leaving according to a density-dependent function (Swartzman 1984) and divides them into male and female groups assuming a 1:1 sex ratio.

$$xpup_j(t+0.25) = \frac{1}{2}k_1(1.0 - k_2e^{-k_3xpup(t)}) \quad (6)$$

Parameters are k_1 , k_2 , and k_3 in this density-dependent relationship. The seal entanglement rate is assumed to be age, sex, and time specific. A year-specific driving variable, $zprop(t)$, the proportion of entangled subadult males observed in the harvest, is multiplied by an age-specific and sex-specific variable:

$$gent_1(t) = zprop(t)k_4 \cdot e^{-k_5 \cdot i} \quad (7)$$

Here k_4 is the ratio of entanglement rate for pups to the proportion of subadult males in the harvest entangled in small debris, and k_5 is a parameter controlling the age susceptibility of seals to entanglement. Entanglement is represented as an exponential function of age, with youngest seals being most susceptible. The parameter k_5 controls the rate of decline of entanglement susceptibility with age. Setting $k_5 = 0.0$ makes all ages equally susceptible. As a way of simplifying sensitivity analysis, this function was used to represent age changes in susceptibility to entanglement by a single parameter, rather than a vector of parameters. The entanglement rate of age i seals in large debris is equal to $k_6 \cdot gent_1$. The model assumes that seals entangled in large debris die rapidly enough

for the mortality rate to be equal to the entanglement rate. Thus, the entanglement rate in large debris is equated to a mortality rate $gmrtlg_{1j}$ (as shown in equations 1 and 2). The parameter k_6 is the ratio of entanglement in large versus small debris.

The mortality rate of seals entangled in small debris $kmrtent_{1j}$ was for convenience also modeled as a function of age and sex. As with entanglement rate, an exponential function was used because it gives flexibility in the change of entanglement mortality with age. The equation is:

$$kmrtent_{1j} = kent1_j e^{-kent2_j \cdot i} \quad (8)$$

Here $kent2_j$ is a sex-specific parameter for changes in the mortality rate of entangled seals with age. It is analogous to k_5 . The mortality rate for age 0 seals (i.e., pups) is $kent1_j$.

Model Parameters

Parameter values used in this model are given in Table 2, along with sources of data. A calibration process was used to improve the fit between the model and data. It consisted of changing selected parameters to produce agreement with pup counts on the Pribilof Islands. During calibration, parameters were constrained to be changed only within "reasonable" limits ("reasonable" depending upon the accuracy of the parameter estimate).

The parameters k_1 , k_2 , and k_3 were estimated using a nonlinear regression based on equation (6) of estimates of pups born against estimates of pups leaving the Pribilof Islands. The regression gave estimates of $k_1 = 1.06 \times 10^6$, $k_2 = 1.007$, and $k_3 = 1.04 \times 10^{-6}$. When the model was run with best estimates for these and other parameters (see Table 2), the fit from 1911 to 1950, the period of population growth, was very poor. We had ascertained earlier (Swartzman 1984) that the population behavior was very sensitive to these parameters (i.e., k_1 , k_2 , and k_3). This being so, we used bootstrap resampling to obtain estimates of the variance of each parameter (by redoing the regression with different resamples) and then "searched" the parameter space (1,000 Monte Carlo runs) to see which combinations of parameter values provided the best fit to the data during the population growth period. This experiment produced the values for k_1 , k_2 , and k_3 given in Table 2.

Model-Data Comparison

Our initial desire for this model was to have it replicate the female fur seal population abundance. Any model unable to do that must be judged insufficient for investigating the effect of entanglement on fur seal population dynamics. Figure 1 compares the model to pup numbers and bulls (for the model this includes all bulls 7 years or older), which are the only long-term data available. The vertical dashed lines in Figure 1 show (from left to right) the year pelagic sampling of females began, the year entanglement began, the year commercial pelagic harvest ended, and the year all female sampling ended (both scientific sampling and commercial harvests).

Table 2.--Parameter values and data sources for northern fur seal population dynamics model.

Parameter	Definition	Value	Estimate source
k_1	Maximum land pup survival	728451	Regression; Lander (1981) calibration.
k_2	Density-dependent land pup survival	0.982037	Regression; Lander (1981) calibration.
k_3	Density-dependent land survival exponent	1.609502e-6	Regression; Lander (1981) calibration.
k_4	Ratio of age 0 seal entanglement to fraction of entangled subadult males	5	Fowler (1982) calibration.
k_5	Change of entanglement with age	0.35	Calibration.
k_6	Ratio of large to small net entanglement rates	15	Fowler (1984) calibration.
$kent1_1$	Entanglement mortality rate for younger males	0.8	Fowler (1982).
$kent1_2$	Entanglement mortality rate for younger females	0.8	Fowler (1982).
$kent2_1$	Entanglement mortality age effect for males	0.35	Calibration.
$kent2_2$	Entanglement mortality age effect for females	0.35	Calibration.
$kfmrt_2$	Ratio of young female to male mortality rate	0.6	Chapman (1964) calibration.
ks_1	Density-dependent survival to age 2	0.5428	Regression; Lander (1981).
ks_2	Density-dependent sea survival parameter	0.7643	Regression; Lander (1981).
ks_3	Density-dependent sea survival parameter	0.7372	Regression; Lander (1981).
Male seals (age-class-specific)			
$kmrt_{11}$	Seal mortality rate	--	Lander (1981).
$kesc_{11}$	Age-specific escapement from entanglement	0.0008	Calibration.
Female seals (age-class-specific)			
$kmrt_{12}$	Seal mortality rate	--	Lander (1981).
$kesc_{12}$	Age-specific escapement from entanglement	0.0008	Calibration.
$kfec_1$	Age-specific fecundity	--	York and Hartley (1981).

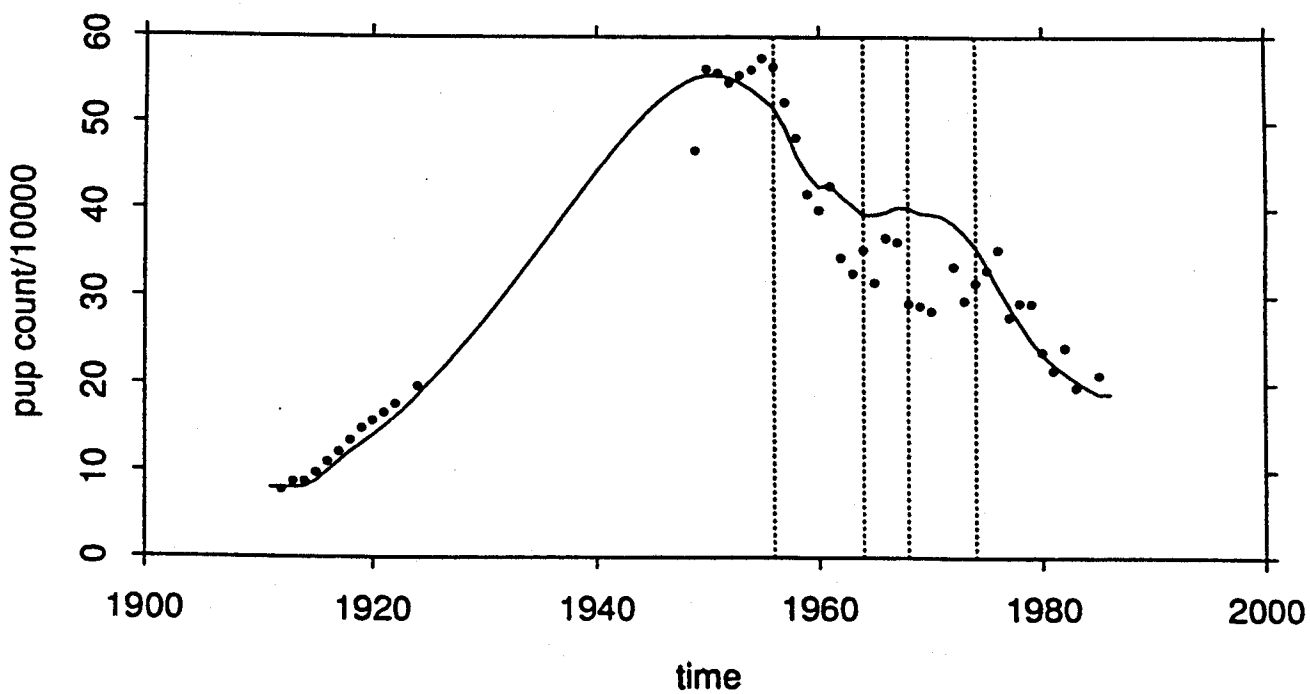
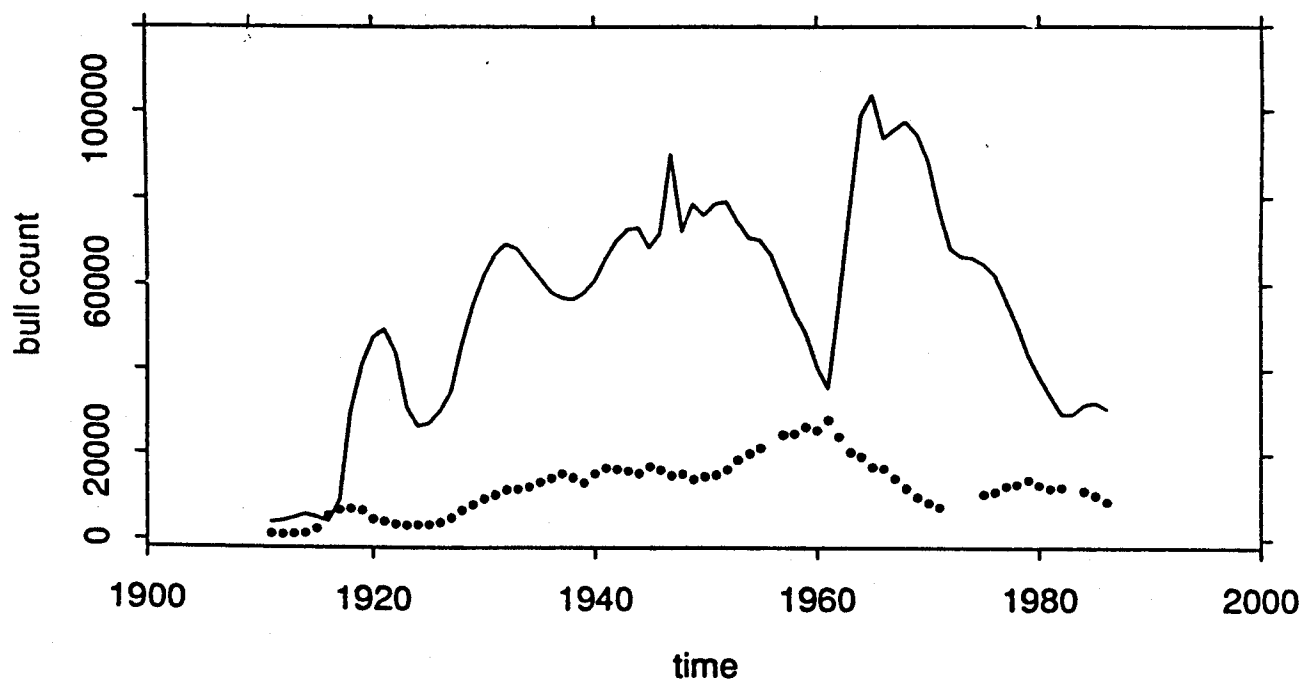


Figure 1.--Model-data comparison between pup and bull counts, 1911-86.

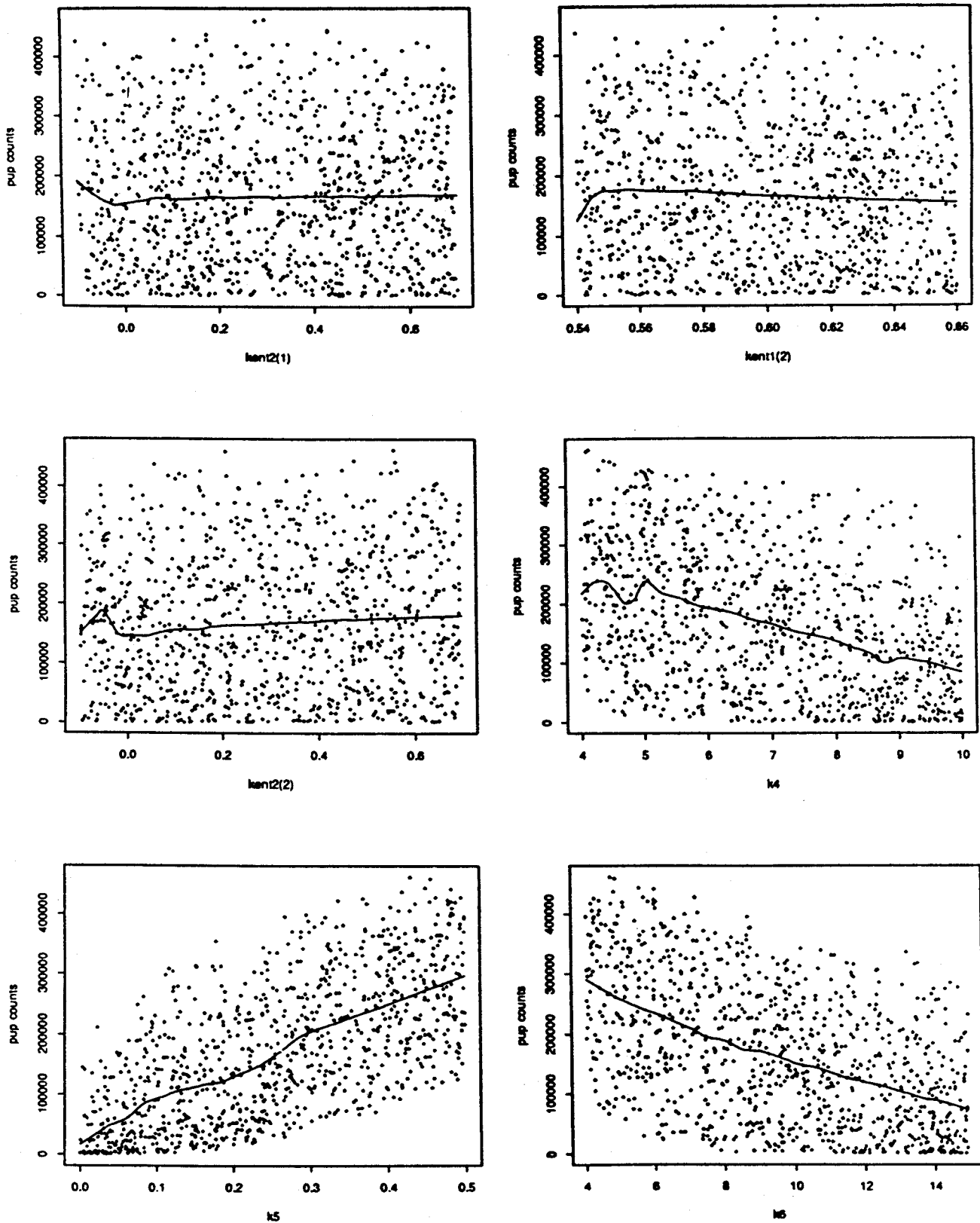


Figure 2.--Sensitivity of 1986 pup counts to entanglement parameters.

Although the fit to the pups appears credible, the bull counts are overestimated by the model. Through many runs of the model adjusting male parameters (the female population is unaffected by these changes), it became apparent that in order for the model not to seriously underestimate the number of bulls during the late 1950's and early 1960's (a period of large subadult male harvests), bull counts in the model during the earlier period need to be significantly higher than reported counts. The possibility that the model was in error was minimized by our checking the calculations and also observing that the bull counts do not appear to respond to marked annual changes in harvests, especially during the 1940's. Possible explanations for this model-data disparity (remember that during this period the model appears to fit pup counts well) are that (1) the actual number of bulls is much higher than the number of territorial bull counted both with females (harem bulls) and as "idle bulls," (2) the pup estimates during this period are in error and pup numbers were actually significantly higher than those obtained by the tagging estimates made in the 1950's (Chapman 1973), or (3) that estimated survival parameters for males are in error.

If the first alternative is true, then many mature bulls, especially the younger ones (e.g., ages 7 and 8), are at sea much of the time and do not show up in the bull counts. If the model's bull predictions are to be believed, there must have been a very large pool of idle bulls in some years that spent either little or no time on land in the Pribilof Islands. Furthermore, the size of this pool has changed over time, being large in the 1930's and 1940's, small around 1960, large again in late 1960's, and now being drastically reduced.

SIMULATING THE EFFECT OF ENTANGLEMENT ON FUR SEAL POPULATIONS

The fit between model and data in Figure 1 was based on a calibration, where entanglement parameters were selected to best fit the population trajectory for pups after 1960. As recorded in Table 2, several of these entanglement parameters are based on limited data and others are simply based on achieving a fit of the model to the data. This is true of the age specificity of both entanglement rate and mortality rate of entangled seals, which has not been studied in the field. The ratio of the rate of seals entangled in large versus small debris is based only on the relative incidence of these two kinds of debris in land and pelagic surveys (Fowler 1987). Also, no studies have been devised to estimate susceptibility of seals to entanglement in debris. Therefore, it is to be expected that our uncertainty about the values of these parameters is great.

To investigate the sensitivity of model predictions of pups and adult males in 1986 (chosen as a measure of model performance that directly relates to the effect of entanglement) to changes in entanglement-related parameters, we performed a Monte Carlo sensitivity experiment. Parameter values were sampled from a uniform distribution over the range of values judged to be reasonable (within our expectation of what the parameter values may be). Our choice is, of course, somewhat subjective. It is to be expected that our range of acceptable parameter values will narrow considerably as a result of this experiment. This method of using a sensi-

tivity study to narrow the tolerance limits on parameter values by choosing combinations giving realistic model behavior was devised by Hornberger and Spear (1981).

Because we are primarily interested in sensitivity of the seal population to entanglement, we restricted our sensitivity study to parameters directly related to entanglement. These include parameters k_4 - k_6 , relating to the entanglement rate in both large and small debris and the change over age in susceptibility to entanglement; $kent1_j$ and $kent2_j$, the entanglement mortality rates for males and females and how they change with age; and $kesc_{ij}$, the age-specific and sex-specific rates of escape from entanglement. The pup and bull populations predicted by the model in 1986 were used as an output variable for comparing sensitivity runs. From Figure 1 it is seen that the population consistently declines after 1960, the year entanglement mortality begins to take effect, and therefore the 1986 value is a measure of the degree of decline (all sensitivity runs are at the same population level in 1960 because they differ only in entanglement-related parameters).

Initial results indicated that the escape-from-entanglement parameters $kesc_i$ are significantly less influential than the other parameters. Therefore the Monte Carlo runs were restricted to the other seven parameters. Table 3 gives the values and ranges used for each parameter in the sensitivity study. Due to uncertainty about the parameter values, we chose to sample parameter values from uniform distributions.

Ranges of parameters were set as follows: $kent1_j$, the age 0 small debris entanglement mortality rate, was set to a range of 10% on either side of the baseline run estimate value of 0.6. Considerable effort has been devoted to estimating the mortality rate of entangled seals, both through observation of marked entangled animals and by looking at the age distribution of entangled versus unentangled seals in the subadult male harvest (Fowler 1987). As such, a modest range of variability was assumed. Three parameters control the age distribution of entanglement effects. Parameter $kent2_j$ is the exponent controlling the age distribution of entanglement mortality for male ($j = 1$) and female ($j = 2$) seals, and k_5 is the same for entanglement rate (no sex distinction here). Baseline estimate for each of these parameters was 0.35. Little is known about how susceptibility to entanglement changes with age, except that significantly more young seals are observed entangled in debris on the Pribilof Islands. Having k_5 of 0.35 makes 0-age seals 20 times as susceptible to entanglement as 8-year-olds. A range of 0 (no difference in age susceptibility) to 0.5 (ratio of 55 in age 0 to age 8 susceptibility) seemed adequate to cover the plausible range of values. For $kent2$ and $kent4$ a wider range, from -0.1 (older animals die more rapidly when entangled) to 0.7, was chosen, reflecting our having no data on how long animals at different ages survive when entangled.

The last two sensitivity parameters, k_4 and k_6 , are not well known. For k_6 , the ratio of entanglement rates in large to small debris, Fowler (1984) estimated a value of 5 based on the ratio of large to small debris in beach surveys on Amchitka (Merrell 1980) and the Pribilof Islands.

Table 3.--Parameter values and ranges for sensitivity analysis.

Parameter	Monte Carlo distribution	Best estimate
$kent1_1$	U(0.54, 0.66)	0.6
$kent2_1$	U(-0.1, 0.7)	0.35
$kent1_2$	U(0.54, 0.66)	0.6
$kent2_2$	U(-0.1, 0.7)	0.35
k_4	U(4, 10)	5
k_6	U(4, 15)	15

However, our baseline estimate, which resulted in a reasonable model-data fit, was 15. We therefore chose a range of 4 to 15, putting our estimate at the high end in deference to Fowler's more data-based measure. Parameter k_4 represents the ratio of entanglement rate in small debris for 0-age seals to the fraction of observed subadult male seals entangled on the Pribilof Islands. The latter is the only data-based time series on annual entanglement available. Estimating k_4 is like trying to assess the size of an iceberg from the part above water. There is a lot unknown below the surface. For a range of values, we blanketed our baseline estimate, 5, by 4 and 10, the relatively high lower bound being due to results of preliminary experiments with the model that showed low values of k_4 leading to an overprediction of pup abundance (too weak an effect of entanglement). One caveat of the calibration approach to parameter estimation in this case is that we are assuming entanglement to be the sole cause of the additional mortality since 1960. If, in fact, there are other yet-undiscovered causes, then entanglement parameter values estimated here would be biased. This is to be borne in mind during the discussion of the sensitivity analysis, which examines parameter ranges that lead to realistic behavior, assuming that all sources of mortality are accounted for in the model (either through harvest, sampling, entanglement, or natural mortality, or through density-dependent juvenile survival).

SENSITIVITY ANALYSIS RESULTS

Figure 2 shows results of the sensitivity study for six of the parameters plotted against pup numbers. Results are omitted for $kent1_1$, which is similar to $kent1_2$. A smooth using supersmoother (Friedman and Stuetzle 1982) was fit to each plot. The three entanglement susceptibility parameters appear to have a stronger effect on 1986 pup numbers than the entanglement mortality parameters. Low values of k_5 (no age-specific or weak age-specific entanglement susceptibility) appear to lead to low 1986 pup predictions.

To get a sense of which parameter combinations led to realistic predictions, we extracted those runs (of the 1,000 runs made) that gave 1986 pup estimates between 170,000 and 200,000 (the baseline run gave 189,000 pups in 1986, close to the pup count estimate for that year).

Figure 3 illustrates which combinations of parameters give realistic model behavior. The first three plots show the entanglement rate parameters against each other for pup numbers within this range, including a super-smooth fit of the resulting scatterplots. The next three plots show each of these parameters' values against pup numbers (for the latter restricted between 170,000 and 200,000 pups). These indicate that over the range of "realistic" pup numbers, none of these parameters has a significant effect on pup numbers (there is no significant slope to the smooths). From the first three plots we deduce that k_4 and k_5 are inversely related to each other, and k_5 and k_6 are inversely related to each other. This can be interpreted to mean that there cannot be a low value of k_5 and a high value of k_4 (and contrariwise a high value of k_5 precludes a low value of k_4). This information is useful because it sets limits of parameter combinations leading to realistic behavior. Furthermore, if more information becomes available concerning any one of these parameters it further delimits the possible values of the other parameters. For example, if observation of entangled animals at sea would indicate that seals are more susceptible to entanglement in large debris than the ratio of large to small debris in beach surveys would indicate (implying a value of k_6 near the upper end of the 4 to 15 range used here), then the entanglement rate needs to be age-specific, with older animals significantly less susceptible to entanglement than younger animals. Significant benefit for research direction can be derived from these results, because they suggest that improved estimates of entanglement rates in small debris can be obtained by seemingly unrelated (and potentially less expensive) than the studies such as finding out about age-related susceptibility to entanglement.

Another interesting result of this sensitivity study is that entanglement rates are much more important for population survival than is entanglement mortality. Another way of saying this is that the rate at which seals enter the entangled animal pool is more important to long-term population trends than the rate at which they die once they are in it. Assuming mortality rate much larger than the rate of escape from entanglement assures that most animals entering the entangled pool will die before they can leave their mark on future generations through reproduction.

ENTANGLEMENT QUESTIONS AND RESEARCH

The model can be used to explore recovery scenarios such as how the population would respond to removing entanglement or reducing it. However, such an exercise is unnecessary. Except for short-term effects, the response to removing entanglement can be observed in Figure 1 in the model's pup counts during the upward cycle starting around 1923. Intermediate entanglement rates would result in less rapid recovery rates. The actual rate of recovery depends on the specific entanglement rates and combinations of parameters. These include the entanglement parameters (k_4 , k_5 , k_6) as well as the non-entanglement-related parameters that were calibrated to fit time traces from 1915 to 1960 pup counts.

More important than recovery scenarios are dominant questions suggested by the model concerning entanglement. These are:

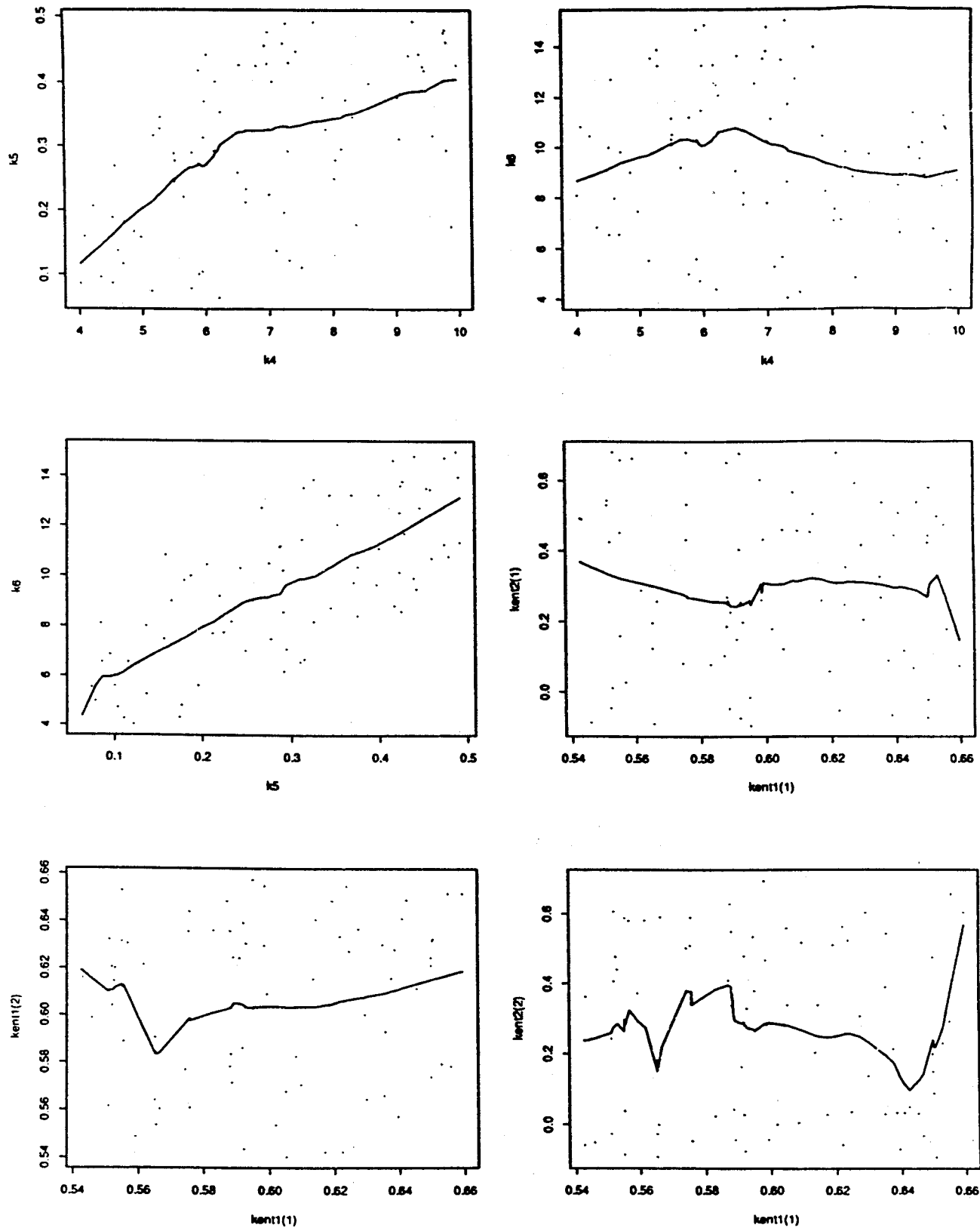


Figure 3.--Parameter interaction for 1986 pups and bulls close to nominal run levels.

1. What is the rate of entanglement in small debris and how is this influenced by debris abundance and distribution?
2. Are there age differences in susceptibility to entanglement and if so how can we measure them?
3. How relatively susceptible are seals to entanglement in large versus small debris?

This modeling exercise has demonstrated that, within the degree of uncertainty that we can answer the above three questions, entanglement is a plausible explanation for the decline in Pribilof Island fur seal populations since the late 1960's. The model's representation of male abundance has raised some questions about using the idle and harem bull counts as an index of total adult male abundance. At the very least, large fluctuations in male harvest are not reflected by subsequent appropriately large changes in bull counts. At the most, a variable, potentially large fraction of the mature males either may not be resident on the Pribilof Islands during the summer or may be resident for only part of the summer. Finally, the modeling has defined the research questions that can help reduce uncertainty about the possible past and future effects of entanglement on seal populations. The first question, about entanglement susceptibility, requires increased (and preferably simultaneous) observation of entangled seals and debris at sea and the development of a debris encounter probability estimate and an estimate of the probability of a seal's being entangled given that it has encountered debris (Ribic and Swartzman 1990). The second question requires taking a closer look at the age-sex distribution of entangled seals on land and perhaps conducting tank experiments on a larger scale than previously done. The third question requires observation of entangled animals at sea and development of statistical methods for estimation based on very infrequent encounters. Research around both the first and third questions may benefit from additional models designed to test various assumptions made in doing the estimates. For example, a Monte Carlo seal-debris encounter model, coupled with further transect observations might help clarify what the probability of entanglement is, given an encounter at sea.

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HISTOLOGICAL OBSERVATION OF DAMAGE TO DERMAL TISSUE
OF FUR SEAL CAUSED BY NET ENTANGLEMENT

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ABSTRACT

In 1984 and 1985, experimental studies on the damage to northern fur seals by net entanglement were carried out in Izu-Mito Sea Paradise, an aquarium where fur seals are kept for the National Research Institute of Far Seas Fisheries, Fisheries Agency, the Government of Japan (formerly the Far Seas Fisheries Research Laboratory, Fisheries Agency of Japan).

Two adult female fur seals, captured off Sanriku, northern Japan, in March 1980, were experimentally entangled in fishing net fragments in late January 1984. One seal, No. 1-7, 123 cm in body length and 39.5 kg in body weight, and an estimated 10 years of age, was entangled around the neck with a net fragment weighing 200 g. The second seal, No. 1-8, 113 cm in body length and 36.0 kg in body weight, with an estimated age of 10 years, was entangled with net fragments 300 g in weight. Both seals were kept in the same environment.

Seal No. 1-8 died in September 1984 after 226 days of entanglement, having suffered traumatic damage to pelage and skin. Net fragments were removed from No. 1-7 in March 1985. Damage to skin and pelage was not observed even after 14 months on entanglement, although the state of entanglement was similar to No. 1-8. She died in February 1986. Abnormality was not observed in the skin at the time of death. Cause of death was acute pneumonia in both cases. When they were alive, No. 1-7 was in good health and No. 1-8 was slightly unwell.

INTRODUCTION

Fur seal are known to have died from entanglement in fishing net fragments and packing bands (Scordino 1985). It was pointed out that net entanglement may constitute one of the major factors for the decrease in the Pribilof Islands populations of the northern fur seal, *Callorhinus ursinus*. In order to understand the actual state and effects of entanglement, the National Research Institute of Far Seas Fisheries, Fisheries Agency, the Government of Japan (formerly the Far Seas Fisheries Research Laboratory, Fisheries Agency of Japan) conducted field surveys of fur seals and marine debris on breeding islands and at sea. Also, experimental studies were carried out concerning the process of entangling and the effects of entanglement on the behavior of fur seals, both of which were difficult to observe in a natural environment. Little has yet been reported about traumatic wounds to the skin of fur seals caused by net entanglement. In this study, macroscopic observation was conducted on damage inflicted by net entanglement to the pelage and skin of fur seals kept in captivity. Post-mortem histological examination was also made of the lesions of the dermal tissues.

MATERIALS AND METHODS

The experiment was conducted for about 14 months from 10 January 1984 to 31 March 1985 in a breeding facility of the Izu-Mito Sea Paradise, an aquarium in Numazu, central Japan. Two female northern fur seals, with identification numbers 1-7 and 1-8 and an estimated age of 10 years, were used. These individuals were entangled with net fragments during an experiment on the mechanism of entanglement. Seal No. 1-7 was entangled in trawl net fragments of 200 g on 20 January 1984, and No. 1-8 was entangled in a trawl net fragment weighing 100 g on 28 January 1984 and in another fragment of 200 g on 29 January 1984. Nets were removed from No. 1-7 at the end of the experiment on 31 March 1985. Behavior of the two seals and damage to pelage were observed every day in the morning and evening. The net fragments used in the experiment were commercial trawl nets made of polyethylene, with a twine size of 3.4 mm and a mesh size of 24 cm. The nets were cut into 100- and 200-g pieces.

The two individuals were kept in an open breeding facility (Fig. 1) from 10 January to 10 March 1984; afterward they were brought to an indoor breeding pool 1.6 m wide, 2.4 m long, and 1.0 m deep. No landing place was provided in the pool, in order to prevent resting on land except for the breeding season, in imitation of pelagic life. The experimental animals were fed with defrosted mackerel each day at 1000 and 1630. Average daily food consumption of No. 1-7 was 3.8 kg/day (0-5.5 kg/day) and that of No. 1-8 was 2.8 kg/day (0.4-4.0 kg/day). Body weight of No. 1-7 remained almost constant during the entire experiment period, while that of No. 1-8 declined near the time of death.

Necropsy was conducted immediately after the death of the entangled animals for macroscopic and histological inspection of the skin lesions. Histological samples of dermal tissue were fixed with 20% formaldehyde, embedded in paraffin, and cut into sections 4 microns thick with a sliding

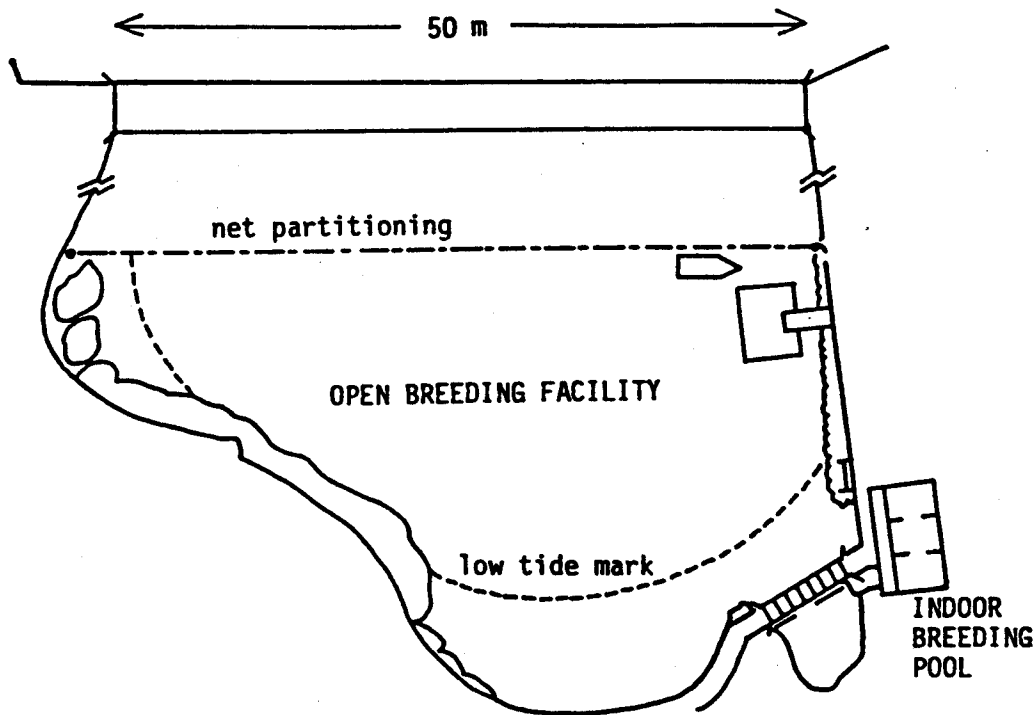


Figure 1.--The breeding facilities at Izu-Mito Sea Paradise.

microtome. The sections were dyed using hematoxylin eosin staining or Masson's method.

Atmospheric temperature during the period ranged from -5° to 30.1°C , water temperature ranged from 12.0° to 28.3°C , and relative humidity ranged from 33 to 92%. Details of the fur seals used in the experiment are given in Table 1.

RESULTS

Behavior of No. 1-7 and Description of Net Entanglement

This individual was in quite good health. On 20 January 1984, she was entangled around the neck by a bundle of net made into a collar. There was a space between the net collar and the neck into which one finger could be inserted. No damage to the entangled part was observed even after 14 months. Net fragments were removed at the end of the experiment on 31 March 1984. The individual remained in good health with a good appetite, finally dying on 25 February 1986, 11 months after the removal of the nets. Post-mortem examination revealed small whitish nodules about 5 mm in size scattered throughout the lungs. No pus was found in the nodules. The cause of death was diagnosed as acute pneumonia. The cervical region where the net had been entangled was also inspected, but no anomaly was recognized.

Table 1.--Details of the two female northern fur seals used in this study.

Chronology of the experiment	Seal No. 1-7	Seal No. 1-8
Capture		
Date	4 March 1980	7 March 1980
Location	Lat. 36°40'N long. 141°27'E	Lat. 36°30'N long. 141°15'E
Transport to aquarium		
Date	8 March 1980	8 March 1980
Body weight (kg)	29.0	30.5
Beginning of experiment		
Date	10 January 1984	10 January 1984
Estimated age	10	10
Body length (cm)	123	113
Body weight (kg)	39.5	36.0
Net entanglement		
Starting date	20 January 1984	28 January 1984
Weight of nets (g)	200	100
		29 January 1984 200
Date of removal	31 March 1985 ^a	10 September 1984 ^b
Death		
Date	25 February 1986	10 September 1984
Body length (cm)	120.5	114.5
Body weight (kg)	34.5	30.5
Cause of death	Acute pneumonia	Acute pneumonia

^aExperiment ended.

^bSeal died.

Behavior of No. 1-8 and Description of Net Entanglement

Judging from daily behavior and feeding activities, this individual was considered in somewhat poor health even before it was entangled in nets. Two net fragments were placed around its neck on 28 and 29 January 1984.

The entangled nets made up a collar consisting of 17 meshes in a bundle (Figs. 2 and 3). The collar was 14 cm in inner diameters, 44 cm in inner circumference, 100 cm in outer circumference, and 7 to 9 cm thick. No remarkable change in behavior was observed after net entanglement. On the morning of 6 June 1984, she delivered a female pup, 65 cm long and 4.2 kg in weight, but the pup was dead when it was discovered. Anatomical inspection revealed the cause of the pup's death to be drowning. Seal No.

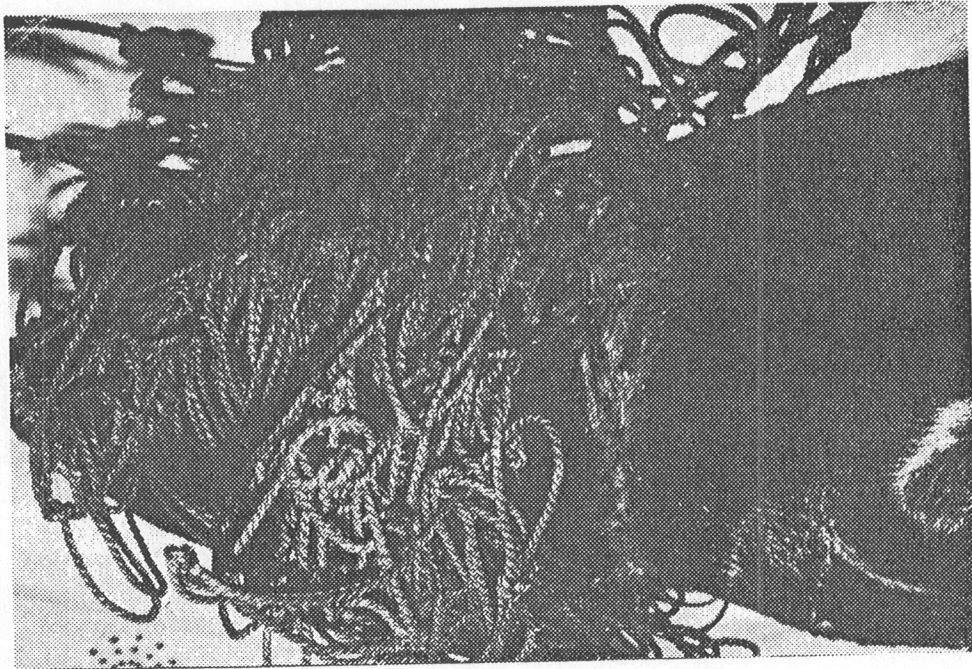


Figure 2.--State of entanglement of fur seal No. 1-8.

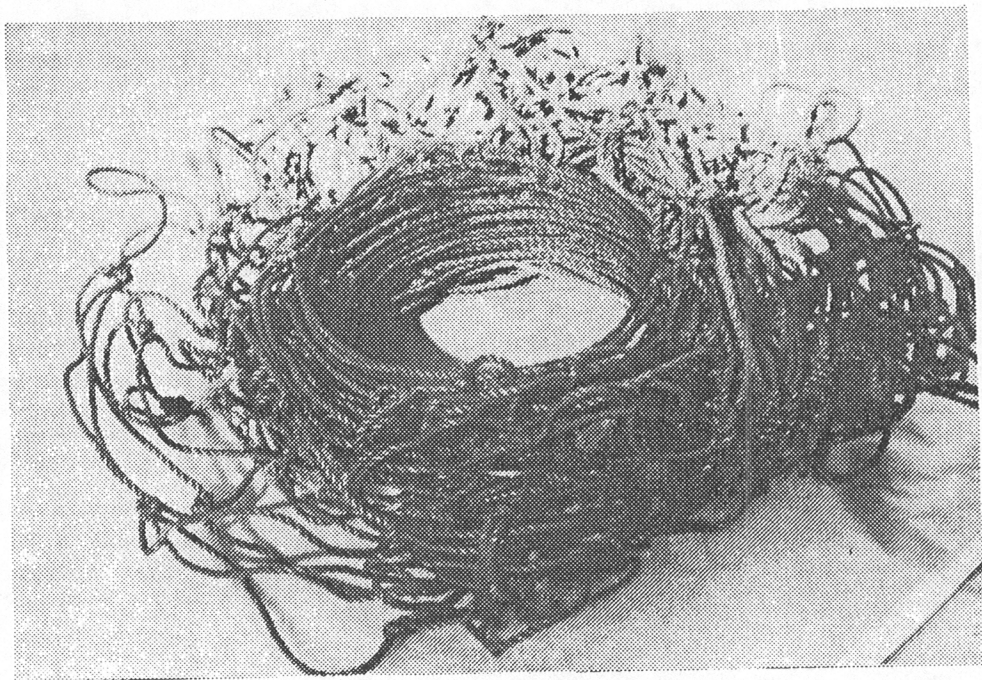


Figure 3.--Collar of entangled net on fur seal No. 1-8.

1-8 ate poorly from 4 September 1984 and died on 10 September, 226 days (7.5 months) after the initiation of net entanglement. Macroscopic and histological observations of the entangled part are described below.

Diagnosis of the Lesions on No. 1-8

The nutritive condition of the body as a whole was moderate. No external scars were observed except around the neck. In macroscopic observation, hair was worn or lost all around the neck, and the epidermis was exposed in some parts. Several wounds due to abrasion by the attached nets were conspicuous: one large scar 1-3 cm wide and 5 cm long on the right side of the neck and two other wounds of 1-2 cm at the scruff. Cross sections of the lesions revealed hardened hyperdermis and a thin muscle layer. The thick adipose tissue consisted of yellow and white parts (Figs. 4 and 5).

In the histological examination, sections of the injured dermal tissue showed degeneration and loss of hair follicles and hair matrices, and degeneration of the hair itself. Connective tissues were partially worn out and necrotized. Proliferation of collagen fiber was observed in the peripheral connective tissues. In these parts, degenerated inner membranes, supposedly derived from venous sinus, were also observed. Rupture and degeneration of muscle layers were distinctive and a part of the connective tissues had been replaced. Congestion and edema of venous ducts were conspicuous and their inner membrane revealed degeneration. However, cell infiltration and inflammatory reactions were indistinctive (Figs. 6 and 7).

As for visceral organs, hyperemia was observed in the lungs and accumulation of blood was conspicuous in the heart. Other visceral organs showed no remarkable change. The cause of death was presumed to be acute pneumonia.

DISCUSSION

No injury occurred around the neck of No. 1-7, entangled in 200 g of fishing nets for 14 months. Seal No. 1-8, entangled at the neck in 300 g of nets for 7.5 months, suffered abrasion of hair and skin. These differences seemed to have been caused by such factors as physical condition, and the amount and tightness of the entangled nets. After anatomical inspection, the cause of death for both individuals was diagnosed as acute pneumonia. As a future task, it will be necessary to examine bacterial infections from the wounds caused by entanglement.

As fur seals used in this experiment were adult, no increase in body weight was observed during the experiment period of 7.5 to 14 months. But it can be assumed that when a young, growing animal becomes entangled in fishing nets, even if the net fragment is small and loose at first, it will gradually become tighter as the animal grows, causing serious damage and possibly death.



Figure 4.--Abraded wound in the pelage and skin of No. 1-8.

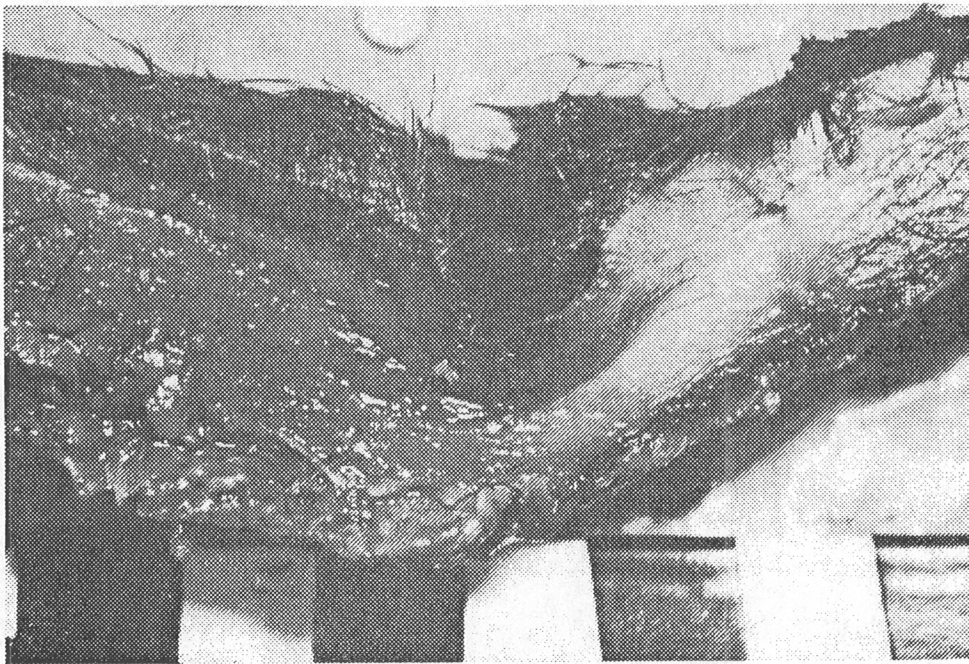


Figure 5.--Cross section of dermal tissues through the lesion on the neck of No. 1-8.

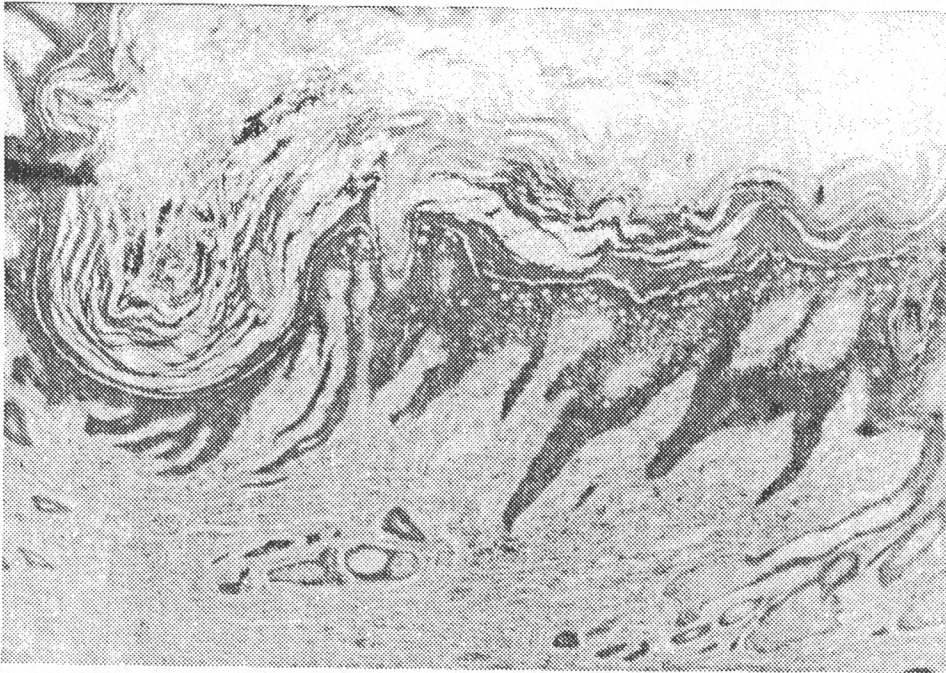


Figure 6.--Section through a lesion in the cervical region of No. 1-8 (hematoxylin eosin staining $\times 40$).

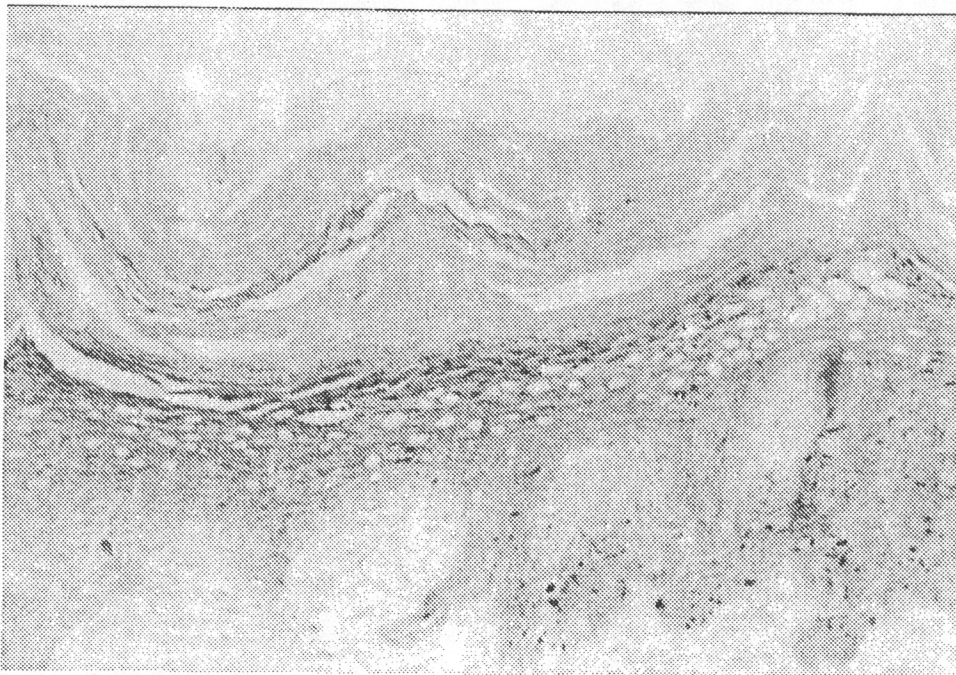


Figure 7.--Section through a lesion in the cervical region of No. 1-8 (Masson's staining $\times 100$).

ACKNOWLEDGMENTS

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RECENT ENTANGLEMENTS OF HAWAIIAN
MONK SEALS IN MARINE DEBRIS

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ABSTRACT

During field studies on the Hawaiian monk seal, *Monachus schauinslandi*, in 1985-88, 34 incidents of entanglement in marine debris were observed, including 4 known deaths, injuries to 4 seals, and recent neck scars on 2 seals. The overall entanglement rate increased eightfold, from 0.06 incidents per 100 camp days per 100 seals in 1985 to 0.48 incidents per 100 camp days per 100 seals in 1988. This increase was probably caused by increased amounts of marine debris on and around the islands where seals haul out. Weaned pups were entangled at a higher rate than their proportion in the population, while adults were entangled at a lower rate. Entanglement rates since 1981, when corrected for each island's population size, were highest at Lisianski Island: 4.44 incidents per 100 camp days per 100 seals. Lowest rates were at French Frigate Shoals: 0.37 incidents per 100 camp days per 100 seals.

INTRODUCTION

In the past 25 years, durable and resilient plastic materials have replaced natural fibers in the maritime industry. Polypropylene and nylon nets have replaced antiquated and once prevalent tarred cotton webbing, and various plastic lines are now used in place of manila or other natural hemp fiber (Pruter 1987). This use of persistent plastics has been accompanied by an increase in the impact of lost or discarded materials on wildlife in the marine environment.

Pinnipeds in particular are susceptible to entanglement in marine debris. Entanglements of the northern fur seal, *Callorhinus ursinus*, are well documented (Fowler 1982; Scordino and Fisher 1983; Scordino 1985) and appear to have contributed to a population decline in this species during 1976-81 (Fowler 1985, 1987). Although other pinnipeds may be entangled less often, the list of species known to have become entangled is large: Fowler (1988) recently stated that 16 of the 34 extant pinniped species (47%) are known to have become entangled in marine debris.

Entanglements of the Hawaiian monk seal, *Monachus schauinslandi*, in marine debris have been observed since 1974 (Balazs 1979; Andre and Ittner 1980; Alcorn 1984; Henderson 1984, 1985). Pups are particularly susceptible (Henderson 1985). Henderson (1985) documented 35 incidents of Hawaiian monk seals entangled in debris through 1984, noting that the number of observed incidents declined following the inception of a program to periodically remove hazardous debris from haul-out beaches at all Northwestern Hawaiian Islands (NWHI). Entanglements have nonetheless continued since 1984 (Alcorn et al. 1988; Johanos and Austin 1988; Johanos and Withrow 1988; Reddy and Griffith 1988; Westlake and Siepmann 1988; Henderson and Finnegan 1990). This report summarizes all of the published and unpublished reports of Hawaiian monk seal entanglements in 1985-88, thereby updating Henderson (1985). This report also examines all entanglements since 1982 for trends in number and location of occurrences, and sizes of affected animals.

METHODS

Staff biologists of the Marine Mammals and Endangered Species Program (MMESP) of the Southwest Fisheries Science Center Honolulu Laboratory, National Marine Fisheries Service, NOAA, conducted field operations from 1982 to 1988 in the NWHI to monitor the Hawaiian monk seal population. Since 1982, most Hawaiian Islands west of Necker Island (i.e., French Frigate Shoals, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway, and Kure Atoll) have been visited and included as study sites. The number of field camp days has varied among the six locations, as has the annual total.

During these studies, all occurrences of entangled or entanglement-scarred seals were recorded. A seal was considered entangled if any part of its body was encircled by debris. Seals resting or asleep on netting or lines were not considered entangled unless their head or body was inside a loop and field personnel thought the animal would not be able to free itself. Seals with entanglement scars, which are distinguishable from scars resulting from other injuries (Henderson 1985), were documented on scar cards and tallied only in the year in which they were first documented to have acquired the scar. Seals with scars were assumed to have become entangled at the island where first observed.

Data on the number of entangled seals per year and per location were converted to number of occurrences per 100 camp days to account for any variation in the length of the field seasons, and incidents recorded while MMESP personnel were absent from an island have been excluded from most analyses. Two such incidents [documented by the U.S. Fish and Wildlife Service (FWS) and the U.S. Coast Guard] occurred in 1985-88 and have been included only in the overall listing and in data on the size of seals that become entangled.

Data prior to 1985 are from Henderson (1985) except for two entanglements in 1984 at Pearl and Hermes Reef. These were inadvertently omitted by Henderson (1985), and are included here:

<u>Seal</u>	<u>Body part</u>	<u>Type of material</u>	<u>Fate of seal</u>	<u>Comments</u>
Juvenile male	Neck	Plastic ring	Unknown	North Is.
Weaned female	Neck, foreflipper	Net	Unknown	North Is.

The relative number of incidents may be affected by the population size of seals, with more entanglements likely among larger populations. Each island's population is relatively discrete and may differ in size from another island's (Johnson et al. 1982). Furthermore, the total population, as indexed by beach counts of nonpups, increased by approximately 24% in 1983-87 (Gilmartin 1988). In the analysis among islands, the number of incidents at any one location was therefore divided by that island's mean 1983-88 beach count of pups and nonpups. To obtain annual entanglement rates, the entanglement total for each year was divided by the summed mean annual beach counts of pups and nonpups at all islands. Data on mean beach counts are incomplete for 1982; therefore, incidents occurring in 1982 were divided by data from 1983. No total beach counts were collected at Lisianski Island in 1988; hence, the 1988 total beach count includes the 1987 total for Lisianski Island.

Because pups have historically been more susceptible to entanglement (Henderson 1985), a separate analysis divided total incidents per year for 1982-88 by the number of pups known to have been born.

RESULTS

Number of Entanglements, 1985-88

Thirty-four entanglements of Hawaiian monk seals occurred from 1985 to 1988 (Table 1). The total included four known deaths, injuries to four animals, and recent neck scars on two seals. The remaining 24 animals were uninjured and were either released from the debris (20) or escaped unassisted (4). Many of the released seals were loosely entangled; it is unknown whether they might have escaped unassisted. Totals by size classes of the affected seals were 12 pups (nursing or weaned), 7 juveniles, 9 subadults, and 6 adults. When data prior to 1985 were included, the number of documented entanglements of Hawaiian monk seals totaled 71.

Entanglements per 100 camp days per 100 seals decreased to a low of 0.06 in 1985 and rose thereafter to a high of 0.48 in 1988, an eightfold increase (Fig. 1). This trend was also evident if incidents were adjusted by the number of pups born annually, with incidents per 100 camp days per 100 pups being 0.23 in 1985 and 1.34 in 1988 (Fig. 2).

Entanglements by Location, 1982-88

The seal population at Lisianski Island experienced the highest rate of entanglement, at 4.44 incidents per 100 camp days per 100 seals since 1982 (Fig. 3). Kure Atoll was next highest (2.23), followed by Pearl and

Table 1.--Summary of entangled and entanglement-scarred Hawaiian monk seals observed in 1985-88.

Year	Location	Size/sex ^a	Incident ^b	Type of material	Fate of seal	Reference ^c	Comments
1985	Kure Atoll	W/M	E, body	Monofilament line	Rescued	Reddy and Griffith 1988	--
		W/F	E, mouth, body	Fish hook and monofilament line	Rescued	Reddy and Griffith 1988	Slight injury from hook.
	French Frigate Shoals (FFS)	J/M	E, neck	Net	Rescued	L. Martin, FWS, pers. commun.	Would have escaped; mesh broke when pulled, Tern Island.
1986	Lisianski Island (LI)	W/F	E, neck	Net and line	Rescued	Westlake and Siepmann 1988	Slight injury.
		S/M	E, muzzle	Line	Rescued	Westlake and Siepmann 1988	Could have escaped.
	Laysan Island	S/M	E, neck,	Line	Rescued	NMFS unpubl. data	--
	FFS	W/M	E, abdomen	Wire	Found dead	Henderson, observation	Wire probably relic of USCG occupation, East Island.
1987	LI	P or W	E, neck	Net	Found dead	Johanos and Withrow 1988	Uncertain if nursing or weaned pup.
		A/F	S, neck	N/A	Alive in 1988	Johanos and Withrow 1988	Seen without scar in 1986.
		W/F	S, neck	N/A	Alive in 1988	Johanos and Withrow 1988	Acquired scar early in 1987.

Table 1.--Continued.

Year	Location	Size/sex ^a	Incident ^b	Type of material	Fate of seal	Reference ^c	Comments
1987	Laysan Island	J/?	E, body	Net	Rescued	NMFS unpubl. data	Net loose.
		S/F	E, neck	Line	Escaped	NMFS unpubl. data	Molting seal, line not tight.
		A/F	E, neck	Line and floats	Escaped	NMFS unpubl. data	--
		J/M	E, neck, shoulders	Line	Rescued	NMFS unpubl. data	Line loose.
		A/F	E, neck	Line	Escaped	NMFS unpubl. data	--
FFS		J/F	E, neck, abdomen	Net and line	Found dead	Henderson, observation	Shark Island.
		S/M	E, abdomen	Line	Rescued	M. Craig, SI, pers. commun.	Line loose, East Island.
		S/F	E, shoulders	Line	Rescued	M. Craig, SI, pers. commun.	East Island.
		S/?	E, neck	Plastic band	No rescue	NMFS unpubl. data	Band cutting into flesh, Whale-Skate Island.
1988	Kure Atoll	A/F	E, neck	Line	Rescued	Henderson and Finegan 1990	Line not tight.
		P/F	E, neck, shoulders	Net and line	Rescued	Henderson and Finegan 1990	On offshore reef, nursing pup, probably would have died.

Table 1.--Continued.

Year	Location	Size/sex ^a	Incident ^b	Type of material	Fate of seal	Reference ^c	Comments
1988	Kure Atoll	W/F	E, neck	Net	Rescued	Henderson and Finnegan 1990	--
		W/F	E, neck	Net	Rescued	Henderson and Finnegan 1990	--
		W/F	E, neck	Net	Rescued	Henderson and Finnegan 1990	On offshore reef, probably would have died, rescued by USCG.
	LI	W/F	E, muzzle	Plastic cup	Rescued	NMFS unpubl. data	--
		W/?	E, body	Net	Found dead	NMFS unpubl. data	Entire body wrapped in large net, decomposed.
	Laysan Island	J/M	E, neck	Plastic ring	Rescued	NMFS unpubl. data	Tight but no injury, probably would have been choked.
		S/F	E, neck	Net	Rescued	NMFS unpubl. data	Cutting deeply into flesh.
		S/M	E, body	Net	Rescued	NMFS unpubl. data	Net loose, seal probably would have escaped.
		S/?	E, abdomen	Line	Rescued	NMFS unpubl. data	--
		A/F	E, muzzle	Net	Rescued	NMFS unpubl. data	Net loose.

Table 1.--Continued.

Year	Location	Size/sex ^a	Incident ^b	Type of material	Fate of seal	Reference ^c	Comments
1988	Laysan Island	A/F	E, abdomen	Net	Rescued	NMFS unpubl. data	Same net as previous incident.
		J/F	E, neck	Net	Rescued	L. Hiruki, UA, pers. commun.	--
	FFS	J/F	E, neck	Plastic screen	Rescued	R. Withrow, WA, pers. commun.	Shark Island.

^aP - nursing pup, W - weaned pup, J - juvenile, S - subadult, A - adult, M - male, F - female, and ? - sex unknown.

^bE - entangled; S - scarred.

^cSI - Smithsonian Institution, FWS - U.S. Fish and Wildlife Service, USCG - U.S. Coast Guard, UA - University of Alberta, and WA - Waikiki Aquarium.

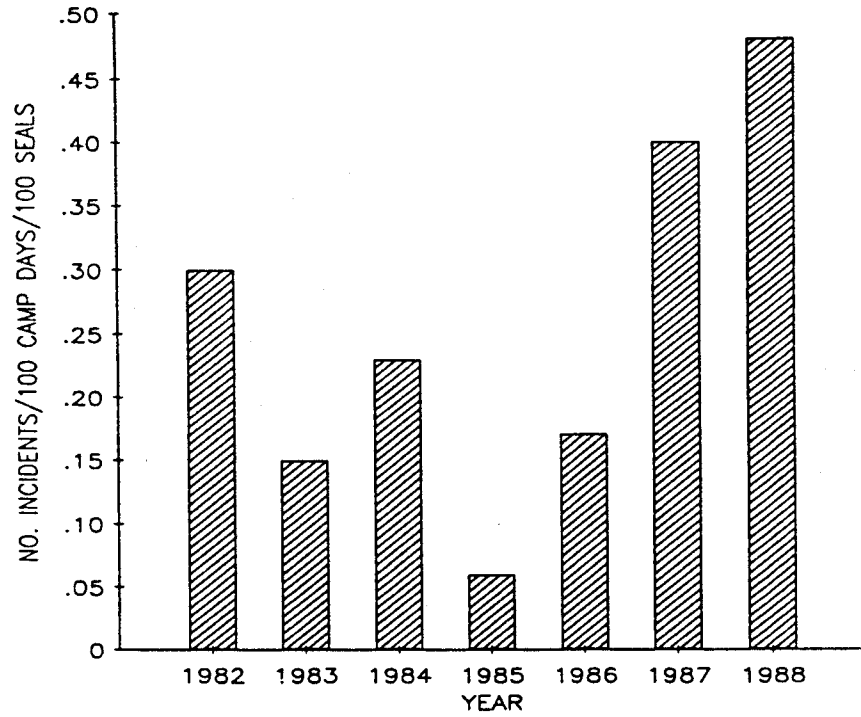


Figure 1.--Annual rates (1982-88) of Hawaiian monk seal entanglement, adjusted for camp days and average beach count (pups and nonpups).

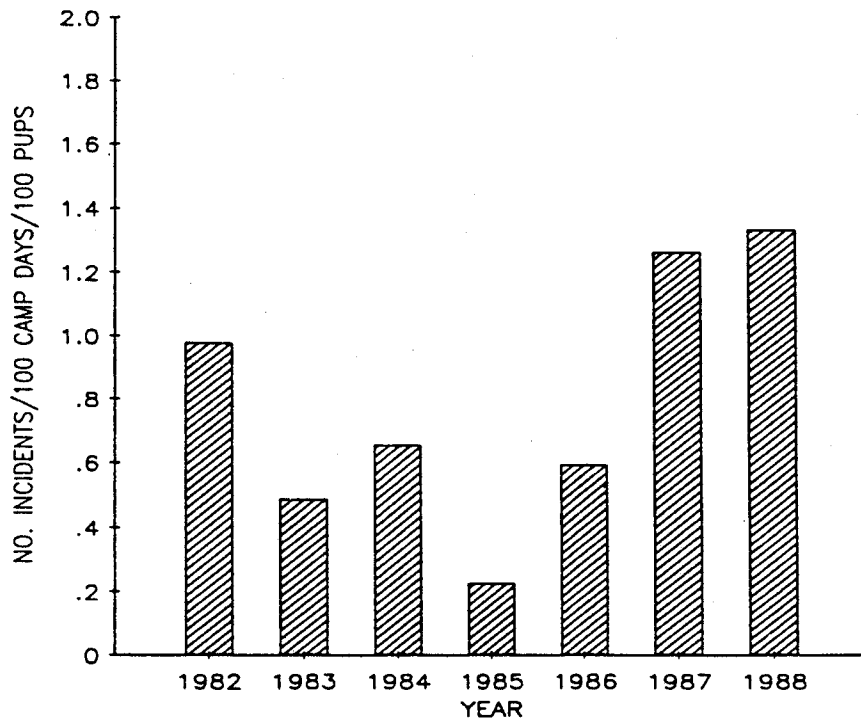


Figure 2.--Annual rates (1982-88) of Hawaiian monk seal entanglement, adjusted for camp days and total pup production.

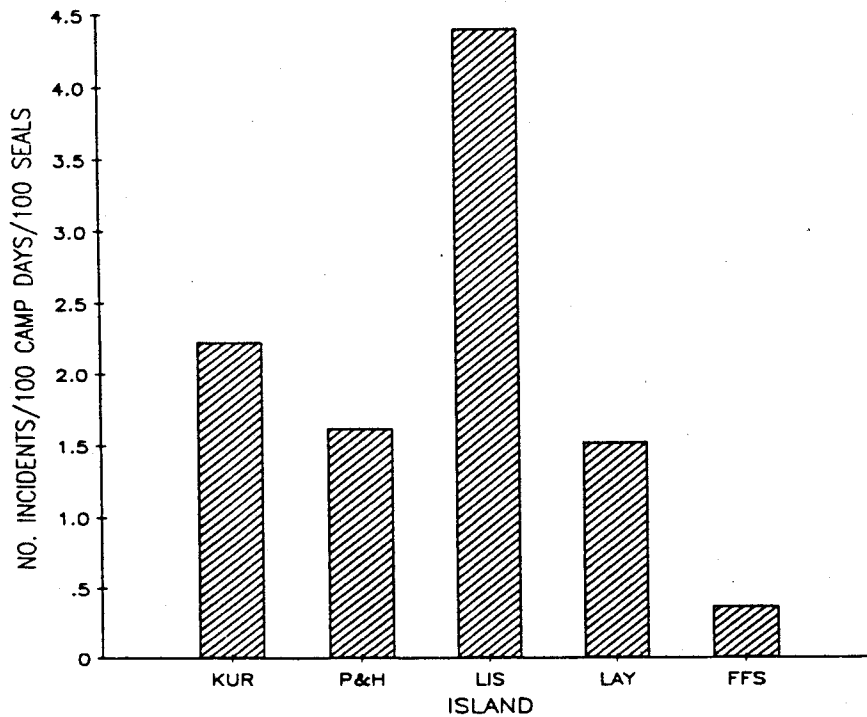


Figure 3.--Location of Hawaiian monk seal entanglement, 1982-88 (Kur = Kure Atoll; P&H = Pearl and Hermes Reef; Lis = Lisianski Island; Lay = Laysan Island; FFS = French Frigate Shoals).

Hermes Reef (1.62), Laysan Island (1.52), and French Frigate Shoals (0.37). One scarred seal has been observed at Midway (Henderson 1985), but MMESP presence there is minimal, beach counts are few, and the seal population is very small (probably 10-20 animals).

Size of Entangled Seals, 1982-88

Pups are most susceptible to entanglement and adults are least susceptible when percent of entanglements is considered in relation to percent of population (Fig. 4). Pups (weaned and nursing) comprise 11.0% of the population (Gerrodette 1985) and yet account for 42.1% of all entanglements from 1982 to 1988. Adults comprise 48.9% of the population and 15.8% of all entanglements. Entanglement rates for juveniles (17.5%) and subadults (24.6%) approximate their population percentages (17.7 and 22.4%, respectively).

Although pups are more susceptible to entanglement, those locations with the most births did not have the most entanglements. Nearly one-third (32.1%) of 1982-88 entanglements occurred at Lisianski Island, whereas only 10.8% of all pups were born there. Conversely, over half (58.8%) of all pups were born at French Frigate Shoals, where 25.0% of the entanglements were documented.

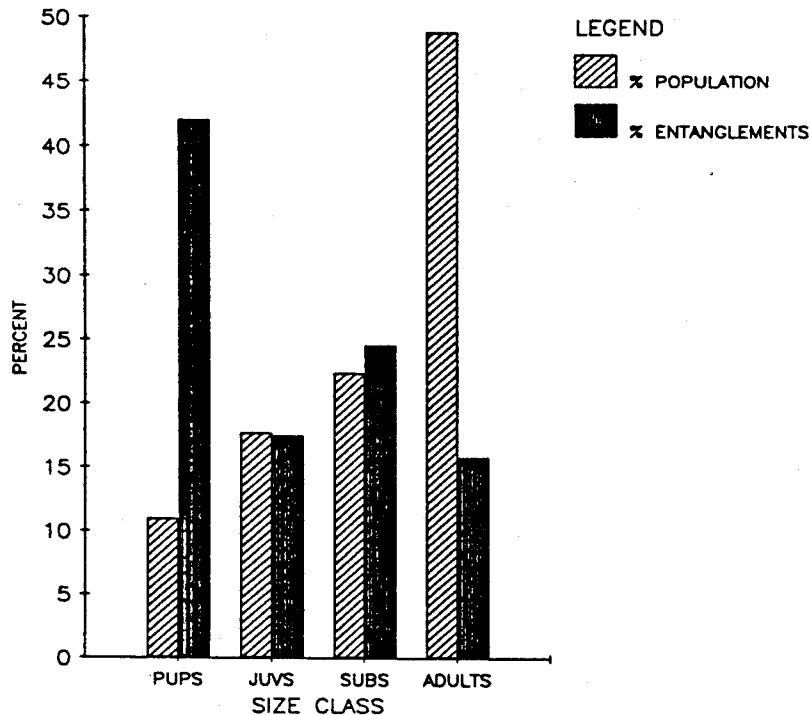


Figure 4.--Size classes of Hawaiian monk seals as percentage of population and percentage of entanglements, 1982-88.

DISCUSSION

The increased incidence of entanglement may result from an increased encounter rate between seals and debris. Such a higher rate could be due to a number of causes: (1) an increase in the seal population, (2) an increase in the relative number of seals in the size class or classes most likely to become entangled, (3) an increased propensity for seals to investigate debris, and (4) an increase in the amount of debris on the islands. Entanglements in 1985-88 have increased more rapidly than any increase in seal population, including the number of pups. In 1985-87, mean beach counts of seals rose approximately 9%, and the number of births also increased by 22% (Gilmartin 1988). However, these increases do not account for the sharp rise in entanglements, which is evident even when adjusted for these factors. No data exist to evaluate whether any recent behavioral changes have occurred among the population to account for increased entanglement.

The amount of debris on beaches in the NWHI has increased in recent years. The number of nets in 1987 alone increased nearly 200% over the 1985-86 average (Henderson unpubl. data). This increase probably has contributed significantly to the rise in seal entanglements. Field biologists routinely remove hazardous debris from the NWHI, an effort credited with reducing seal entanglements (Henderson 1985), yet despite this effort, entanglements have increased. With larger amounts of debris present, more

entanglements may occur (1) during the long periods when personnel are absent from these remote islands, (2) while biologists are present but before the beaches have been cleared, or (3) while seals are at sea.

The higher incidence of entanglement at Lisianski Island may be attributable to spatial coincidence of favorable pupping habitat with areas receiving most of the debris. Pups are born predominately on the island's east side (Johanos and Henderson 1986; Johanos and Kam 1986), which also receives more debris because debris is moved toward the island by trade winds from the northeast (Henderson et al. 1987).

Pups continue to become entangled at a proportionally higher rate than other size classes, a phenomenon that may have several contributing causes (Henderson 1985): (1) entangled pups are most easily observed because they remain near shore for 1-2 months after weaning; (2) pups, unlike older seals, spend proportionally more time in the vicinity of nearshore reefs, which catch and "concentrate" floating debris; (3) weaned pups are not as strong as older seals and are therefore least able to escape from debris; and (4) recently weaned pups are learning to feed and are more likely than nonpups to explore all objects in their novel environment. Bengston et al. (1988) demonstrated experimentally that recently weaned northern fur seal pups readily explore and become entangled in net fragments, and suggested that this behavior could lead to high mortality among fur seals just after weaning.

The periodic presence of biologists in the remote habitat of the Hawaiian monk seal can reduce deaths of seals from entanglement. Of the four mortalities documented here, two could likely have been prevented had personnel been present on the island. Both of these mortalities were at Lisianski Island, a location with sparse coverage by biologists in recent years.

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PINNIPED ENTANGLEMENT IN SYNTHETIC MATERIALS
IN THE SOUTHERN CALIFORNIA BIGHT

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ABSTRACT

The California sea lion, *Zalophus californianus*, the northern fur seal, *Callorhinus ursinus*, the harbor seal, *Phoca vitulina richardsi*, and the northern elephant seal, *Mirounga angustirostris*, that haul out or breed on the southern California Channel Islands, become entangled in synthetic debris at various rates. The percentages of California sea lions entangled, primarily in monofilament gillnet fragments, varied from about 0.08% in 1983 to about 0.16% from 1985 through 1988, while those of northern elephant seals, primarily in packing straps, declined from about 0.15% in 1983 to about 0.10% in 1989. The entanglement rate of harbor seals has varied from 0.0% in 1983-84 to 0.06% in 1986. Entangled northern fur seals have rarely been observed.

Inter- and intraspecific differences in entanglement rates are likely the result of age, sex, and species differences in animal size, diving behavior, and foraging areas. Although entanglement in synthetic materials contributes to mortality of some animals, our studies suggest prevailing entanglement rates have not significantly influenced pinniped demography and population trends in the Southern California Bight.

INTRODUCTION AND METHODS

Pollution of marine environments with nonbiodegradable plastic debris has become an issue of increasing concern during the past several years, especially with regard to entanglement of marine mammals and seabirds in synthetic debris (e.g., see Shomura and Yoshida 1985; Wolfe 1987).

Since 1978 we have studied the incidence of entanglement of pinnipeds in synthetic materials at San Nicolas and San Miguel Islands. We have attempted to distinguish between entanglement in floating marine debris and that resulting from direct interactions of seals and sea lions with commercial fishing and sportfishing operations (Stewart and Yochem 1985, 1987).

Below we summarize the results of our surveys made between October 1986 and March 1989, and incorporate our previous surveys to assess trends in pinniped entanglement in the Southern California Bight since 1983.

We quantified rates of entanglement of the California sea lion, *Zalophus californianus*, the harbor seal, *Phoca vitulina richardsi*, the northern elephant seal, *Mirounga angustirostris*, and the northern fur seal, *Callorhinus ursinus*, using methods described earlier (Stewart and Yochem 1985, 1987). Briefly, we surveyed pinniped populations at San Nicolas Island once each month and those at San Miguel Island periodically whenever we visited there to conduct other research. Using binoculars or a spotting telescope, we systematically examined small groups of pinnipeds on rookeries and hauling grounds and recorded the number examined (by sex and relative age whenever possible), the number entangled (and the type of entangling material), and the number scarred (presumably from prior entanglement); only those animals whose bodies could be seen clearly were sampled during those entanglement surveys.

RESULTS AND DISCUSSION

From March 1988 through February 1989, 30 (0.12%) of 24,731 California sea lions surveyed at San Nicolas and San Miguel Islands were entangled and another 25 (0.10%) were scarred from previous entanglement, slightly fewer than from October 1986 through February 1988 (Tables 1, 2). Slightly more northern elephant seals were entangled but slightly fewer scarred in 1988-89 than in 1986-88 (Tables 1, 3). Relatively few (0.03%) harbor seals were entangled in 1988-89 and none was scarred (Tables 1, 4), and we observed no scarred or entangled northern fur seals (Tables 1, 5). Neither of two Guadalupe fur seal bulls that we observed at San Nicolas Island in summer 1988 was entangled or scarred.

The percentage of California sea lions observed entangled increased from 1983 through 1987 but declined in 1988, while the percentages of entangled northern elephant seals and harbor seals have declined since 1984 (Table 1). The percentages of scarred sea lions and elephant seals observed have remained relatively constant since 1983 and 1984, respectively (Table 2). We have not observed a scarred harbor seal since 1984 nor a scarred northern fur seal since July 1987.

As in previous years (Stewart and Yochem 1985, 1987), the primary material entangling California sea lions in 1988-89 was monofilament gillnet (Table 6); no floats were attached to the entangling monofilament. Since monofilament is negatively buoyant and sinks without the support of floatation devices, we believe that sea lions that were entangled in monofilament became entangled in operational gillnets and were cut out of the nets, leaving some net remaining around the animals' necks. If sea lions (especially young animals) are capable of breaking out of gillnet panels by snapping mesh lines, then some animals may have been entangled in derelict nets (i.e., debris) which were still attached to floats as well as operational, nonderelict nets. Clearly, larger animals, particularly adult males, are capable of breaking free of gillnets once they become entangled (R. DeLong pers. commun.).

Table 1.--Entanglement and scarring rates of pinnipeds in southern California waters by synthetic materials.

Years surveyed	California sea lions	Northern elephant seals	Harbor seals	Northern fur seals
1983-84				
Surveyed	13,174	6,815	1,809	--
% entangled	0.08	0.15	0.00	--
% scarred	0.10	0.09	0.06	--
1985-86				
Surveyed	35,824	17,338	3,342	826
% entangled	0.16	0.16	0.06	0.00
% scarred	0.11	0.20	0.03	0.24
1986-88				
Surveyed	27,733	12,846	3,324	353
% entangled	0.16	0.09	0.03	0.00
% scarred	0.11	0.19	0.00	0.28
1988-89				
Surveyed	24,731	9,775	2,816	422
% entangled	0.12	0.10	0.03	0.00
% scarred	0.10	0.18	0.00	0.00

The observations that we present here, as well as our earlier ones (Stewart and Yochem 1985, 1987), indicate that sea lions become entangled primarily during the first 2 or 3 years of life. Our observations of scarred juveniles and adults indicate that some animals are freed from the entangling material, presumably monofilament, and survive. Others probably die as a result of entanglement either directly through blood loss or indirectly from infection and secondary complications. The magnitude of that mortality is difficult to assess, as many may die at sea, and an insignificant number of tagged sea lions are entangled, preventing an assessment of survival.

For the following discussions we limit the use of the term synthetic marine debris to material other than monofilament. Whether entangling monofilament is obtained during interactions with active fishing gear or from floating derelict nets or net fragments remains difficult to assess. Observations during commercial fishing operations or studies of captive sea lions might clarify whether or not these cases of entanglement are actually related to floating marine debris.

Nevertheless, the increase in numbers of sea lions observed entangled in monofilament in recent years is interesting, considering the restrictions placed in 1983 on the shark and swordfish drift gillnet fishery around the southern California Channel Islands, a fishery that accounted

Table 2.--Entanglement and scarring rates of California sea lions at San Nicolas and San Miguel Islands.

Years surveyed	Adult males	Subadult males	Females/ juveniles	Yearlings	Pups
1983-84					
Surveyed	345	803	7,206	771	4,049
% entangled	0.00	0.12	0.03	0.91	0.02
% scarred	0.58	0.75	0.07	0.00	0.00
1985-86					
Surveyed	1,577	2,272	30,548	1,427	--
% entangled	0.00	0.18	0.15	0.42	--
% scarred	0.44	0.48	0.07	0.00	--
1986-88					
Surveyed	1,384	987	17,619	2,872	4,871
% entangled	0.00	0.00	0.20	0.24	0.02
% scarred	0.14	0.40	0.14	0.00	0.00
1988-89					
Surveyed	710	833	18,670	1,347	3,171
% entangled	0.00	0.12	0.12	0.30	0.13
% scarred	0.70	0.84	0.07	0.00	0.00

for most of the sea lion entanglement and mortality in recent years. Perhaps sea lions became entangled in gillnets north of Point Conception, where the fishery has recently expanded.

In 1988 and early 1989, we were able to confirm that only about 7% of the entangled sea lions observed were entangled in synthetic debris (rubber bands, Table 6). We have no information yet about the survival of sea lions entangled in such debris, and we have not observed any dead sea lions entangled in anything except monofilament gillnet fragments.

All but one entangled northern elephant seal were entangled in synthetic debris (packing bands, Table 6). Elephant seals appear to become entangled during the first 1 or 2 years of life, probably because the circumferences of most packing band debris are too small for the bands to go over the heads of older seals. Scars around the necks of older seals indicate that some seals survive entanglement, although the type of material that entangled those seals is not known. None of the seals that we have observed entangled were tagged, preventing assessment of the influences of various kinds of debris on seals' survival. Five of the adult females that we observed with severely constricting packing bands around their necks gave birth and successfully weaned their pups in 1988.

Since 1983 we have observed only four entangled harbor seals (all juveniles, each with a packing band around its neck), suggesting that they

Table 3.--Entanglement and scarring rates of northern elephant seals at San Nicolas and San Miguel Islands.

Years surveyed	Adult males	Subadult males	Females/ juveniles	Yearlings
1983-84				
Surveyed	1,019	875	4,410	511
% entangled	0.00	0.34	0.07	0.19
% scarred	0.00	0.11	0.07	0.00
1985-86				
Surveyed	1,776	1,485	13,686	391
% entangled	0.00	0.34	0.18	1.02
% scarred	0.28	0.81	0.06	0.51
1986-88				
Surveyed	1,239	1,045	9,802	760
% entangled	0.00	0.00	0.12	0.00
% scarred	0.32	0.77	0.13	0.00
1988-89				
Surveyed	989	658	7,726	402
% entangled	0.00	0.45	0.08	0.25
% scarred	0.81	1.06	0.04	0.00

rarely encounter potentially entangling debris in southern California waters. As it has been speculated that large numbers of harbor seals are incidentally killed each year in gillnet fisheries in southern California, it is surprising that we have seen no harbor seals entangled in gillnets, especially in comparison to the number of California sea lions that are. If young California sea lions that become caught in gillnets are, in fact, capable of breaking out of gillnets, the lack of observations of harbor seals entangled in gillnet fragments may suggest that they are incapable of breaking free. We speculate that such differences may be due to the different modes of propulsion of these two species and consequent differences in potential force generated to permit them to break mesh strands. Harbor seals may then simply die in active or derelict gillnets rather than break free. Clearly, additional observations are needed to sort among these speculations.

Of all cases of pinniped entanglement observed, we can only confirm that 27% were due to marine debris in 1986-88 and 22% in 1988-89, with much of the remainder (perhaps as much as 73%) evidently related to interactions of pinnipeds (especially juvenile sea lions) with commercial gillnet fisheries (see Stewart and Yochem 1987 for additional discussion).

Because relatively few pinnipeds are observed entangled in synthetic material, analysis of trends in entanglement rates (especially debris-related) is difficult. It is clear, however, that relatively large samples

Table 4.--Entanglement and scarring rates of harbor seals at San Nicolas, San Miguel, and Santa Rosa Islands.

Years surveyed	Adults	Immatures
1983-84		
Surveyed	1,445	364
% entangled	0.00	0.00
% scarred	0.07	0.00
1985-86		
Surveyed	2,757	585
% entangled	0.00	0.34
% scarred	0.00	0.17
1986-88		
Surveyed	2,021	1,303
% entangled	0.00	0.08
% scarred	0.00	0.00
1988-89		
Surveyed	1,900	916
% entangled	0.00	0.11
% scarred	0.00	0.00

Table 5.--Entanglement and scarring rates of northern fur seals at San Miguel Island.

Years surveyed	Adult males	Subadult males	Females/ juveniles
1985-86			
Surveyed	58	108	660
% entangled	0.00	0.00	0.00
% scarred	0.00	0.92	0.15
1986-88			
Surveyed	15	63	275
% entangled	0.00	0.00	0.00
% scarred	0.00	1.59	0.00
1988-89			
Surveyed	35	59	328
% entangled	0.00	0.00	0.00
% scarred	0.00	0.00	0.00

Table 6.--Types of synthetic material observed entangling pinnipeds at San Nicolas and San Miguel Islands, 1988-89.

Pinnipeds entangled	Monofilament gillnet	Packing bands	Other debris ^a	Total
California sea lions				
Adult females/juveniles	20	--	2	22
Yearlings	15	--	--	15
Pups	3	--	1	4
Total	38	--	3	41
Northern elephant seals				
Subadult males	--	2	1	3
Adult females	--	6	--	6
Juveniles	--	1	--	1
Yearlings	1	--	1	2
Total	1	9	2	12
Harbor seals				
Adults	--	1	--	1
Total	--	1	--	1

^aIncludes rubber bands, polyfilament rope and line, and items other than trawl or gillnet fragments or nylon monofilament line.

(i.e., systematic observations of large numbers of pinnipeds ashore) are necessary to evaluate properly the true rates of entanglement.

Populations of all pinnipeds have been increasing rapidly in the Southern California Bight during the past two decades (e.g., Stewart 1989; Stewart et al. 1990), indicating that entanglement of pinnipeds in marine debris has had only minor influence on population trends.

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MARINE MAMMAL AND SEA TURTLE ENCOUNTERS WITH MARINE DEBRIS
IN THE NEW YORK BIGHT AND THE NORTHEAST ATLANTIC

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ABSTRACT

The incidence of ingestion of synthetics by, and entanglement of, marine mammals and sea turtles in the New York Bight (1979-88) and in Iceland (1985) was documented and related to the ecology of these animals. Post mortems of 88 cetaceans, 37 pinnipeds, and 116 sea turtles in the New York Bight revealed ingestion of synthetics in 24 animals. Differences were observed among the groups of animals. Synthetics were found in 3 mysticete whales, in 7 odontocete whales (3 delphinids, 3 physterids, and 1 phocoenid), and in 14 sea turtles (10 leatherbacks, *Dermochelys coriacea*, 3 loggerheads, *Caretta caretta*, and 1 green, *Chelonia mydas*). No synthetics were found in the gut of any pinnipeds or in Kemp's ridley turtles, *Lepidochelys kempfi*. Seventy-five individuals were entangled, including 4 mysticetes, 13 odontocetes, and 58 sea turtles. In Iceland, 6 of 82 examined fin whales, *Balaenoptera physalus*, contained ingested synthetics, and 5 of 95 showed signs of previous entanglement. The types of synthetics ingested and the rate of occurrence of both ingestion and entanglement were related to the feeding behavior, timing, and distribution of the species. The results indicate that certain species of marine mammals and sea turtles are more likely to interact with debris than others. In these animals ingestion of synthetics and entanglement appear to be frequent and widespread.

INTRODUCTION

Increased human use of the oceans and inshore waters has resulted in large amounts of man-made materials with which marine organisms come into contact. Organisms interact not only with waste products and floating debris but also with actively used fishing gear. Numerous efforts have been conducted worldwide to assess the amounts (Wehle and Coleman 1983; Bean 1987), types (Carpenter et al. 1972; Dixon and Dixon 1981; Dahlberg and Day 1985; Center for Environmental Education 1987a, 1987b; Henderson et al. 1987), and sources of these materials (Horsman 1982) and their impacts on marine organisms (Shomura and Yoshida 1985; Coe and Bunn 1987; O'Hara 1989). The interactions of marine organisms with these materials, and the

resulting impacts, are better understood when the ecology of the individual species is considered.

Many marine species have global distributions and occur in both populous and remote areas. An abundance and diversity of marine mammals and sea turtles are found in the New York Bight. This is one of the most heavily stressed coastal regions in the world. With New York City at its apex, the bight is a major port for shipping and fishing. This region's coastal population of over 25 million places heavy demands upon the marine environment through activities such as recreational boating, fishing, and dumping of wastes. In contrast, the Arctic region, which supports large populations of marine mammals (Remmert 1980), is one of the few remaining areas in the world where man's influence is still limited. Despite its remoteness, it has been shown that sperm whales in this region were also impacted by marine debris (Martin and Clarke 1986).

The objective of this research was to examine the incidence of ingestion of synthetics by, and entanglement of, different types of marine mammals and sea turtles in the New York Bight and to provide comparisons with whales in Iceland waters.

METHODS

The study was conducted during the period of 1979 through 1988 in the New York Bight and in Iceland during the summer of 1985. Data on ingested materials in the New York area were collected during post mortems of digestive tracts in stranded animals. Only those stranded animals for which reliable necropsies could be performed were included in this study. Animals examined included 37 pinnipeds, 88 cetaceans (19 mysticetes and 69 odontocetes), and 116 sea turtles (Table 1). Data from Iceland were collected by examining the gut contents of 82 fin whales, *Balaenoptera physalus*, at a whaling station in Hvalfjordur during the 1985 season.

Data on entanglement were also collected during the post mortems of both the New York and Iceland specimens (Fig. 1). In New York, a large number of stranded live animals were also examined for evidence of entanglement, e.g., visible scars as reported by Hare and Mead (1987) or actual attached debris, and in Iceland, 13 additional fin whales were examined for entanglement only.

RESULTS

Ingestion of Synthetics

Evidence of ingestion of synthetic materials was found in 24 animals in the New York Bight during this study (Table 2). The frequency of occurrence varied among groups. Synthetics were present in the gut of three individual mysticetes and in seven odontocetes. Among the odontocetes, 3 out of 8 physterids, 3 of 50 delphinids, and 1 of the 9 phocoenids examined contained synthetic materials. There was no evidence of ingestion of synthetics in any of the pinnipeds.

Table 1.--Stranded marine mammals and sea turtles in the New York Bight from 1979 through 1988. A total of 461 live and dead animals were found along the shores or entangled in nets in the water.

Species	Number of individuals
Cetaceans	
<i>Balaenoptera acutorostrata</i>	5
<i>Balaenoptera physalus</i>	9
<i>Delphinapterus leucas</i>	1
<i>Delphinus delphis</i>	15
<i>Eubalaena glacialis</i>	1
<i>Globicephala melaena</i>	14
<i>Grampus griseus</i>	1
<i>Kogia breviceps</i>	5
<i>Lagenorhynchus acutus</i>	4
<i>Megaptera novaeangliae</i>	4
<i>Mesoplodon densirostris</i>	1
<i>Phocoena phocoena</i>	9
<i>Physeter catodon</i>	3
<i>Stenella coeruleoalba</i>	7
<i>Stenella plagiodon</i>	3
<i>Tursiops truncatus</i>	10
<i>Ziphius cavirostris</i>	1
Unidentified	6
Pinnipeds	
<i>Halichoerus grypus</i>	2
<i>Phoca groenlandica</i>	1
<i>Phoca vitulina</i>	34
Sea turtles	
<i>Caretta caretta</i>	103
<i>Chelonia mydas</i>	15
<i>Dermochelys coriacea</i>	85
<i>Lepidochelys kempfi</i>	122
Total	461

Among the sea turtles, varying amounts of synthetics were found in 10 of the 33 leatherbacks, *Dermochelys coriacea*, in 3 of 35 loggerheads, *Caretta caretta*, and in 1 of 4 green turtles, *Chelonia mydas*. Although there were 44 Kemp's ridleys, *Lepidochelys kempfi*, examined in this study, none of these turtles contained synthetics in its gut.

In the Iceland survey during the summer of 1985, plastic material was found in 6 of the 82 fin whales examined.

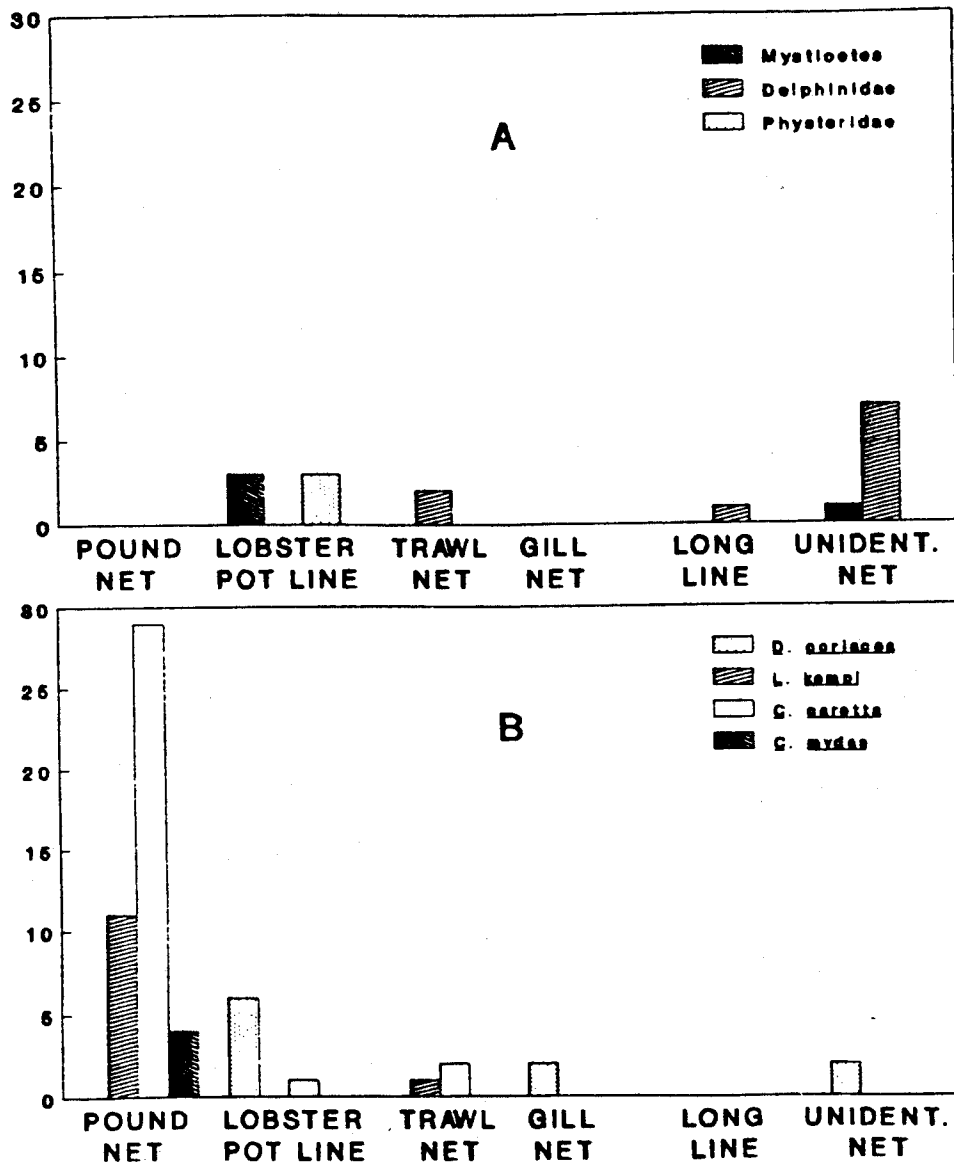


Figure 1.--The incidence of entanglement in different types of gear for marine mammals (A) and sea turtles (B) in the New York Bight from 1979 through 1988.

A wide variety of debris was observed in the stomachs of the animals examined in this study. The various types of debris found in the guts of cetaceans in the New York Bight included plastic toys, cups, polypropylene line, plastic bags, plastic sheets, and some unidentifiable synthetics. Similar materials were found in the Icelandic whales as well. One of these fin whales contained plastic that unfolded to a 1 x 2 m sheet. The most prevalent types of ingested debris observed in cetaceans from both study areas were plastic bags and small pieces of plastic sheeting.

Table 2.--Gut content analysis of marine mammals and sea turtles in the New York Bight from 1979 through 1988 and of fin whales, *Balaenoptera physalus*, from Iceland during the summer of 1985.

Location	Number examined	Number with synthetics
New York Bight		
Cetaceans		
Mysticetes	19	3
Odontocetes		
Delphinidae	50	3
Phocoenidae	9	1
Physteridae	8	3
Ziphiidae	2	0
Pinnipeds		
Phocidae	37	0
Sea turtles		
Dermochelyidae	33	10
Cheloniidae		
<i>Caretta caretta</i>	35	3
<i>Chelonia mydas</i>	4	1
<i>Lepidochelys kempfi</i>	44	0
Total		24
Iceland		
Cetaceans		
Mysticetes		
<i>Balaenoptera physalus</i>	82	6

Various lengths of monofilament line, small pieces of different colored plastic, and numerous small polystyrene balls had been ingested by sea turtles. Most of the synthetic material in sea turtles, however, was clear, thin plastic. In some instances entire plastic bags were present, and these were the predominant synthetic material found in leatherback turtles.

For several stranded animals there was strong evidence that ingestion of synthetics was contributory or causative of death. In one pygmy sperm whale, *Kogia breviceps*, a hard, black plastic ball had completely blocked the pyloric valve. The surrounding tissue was hemorrhagic and there was

extensive necrosis. This animal was also severely emaciated upon death. Another whale, a pregnant sperm whale, *Physeter catadon*, was found with approximately 300 m of polypropylene line wrapped around its jaw and extending into the stomach. The esophagus and stomach were hemorrhagic and the lower jaw was gangrenous at the time of death. Five leatherback turtles had a large bolus of plastic occluding their digestive tracts. One such bolus was made up of 15 quart-size plastic bags and was blocking the pyloric opening.

Entanglements

From 1979 to 1988 there were a total of 75 individuals in the New York Bight that exhibited signs of entanglement with either debris or inactive or active fishing gear (Fig. 1). These individuals included 4 mysticetes, 13 odontocetes, and 58 sea turtles. No pinnipeds in this study were entangled in gear or debris. In Iceland, 5 of the 95 fin whales examined showed signs of previous entanglement.

Types of entanglement varied among groups of animals (Fig. 1). Three of the four mysticetes were entangled in lines from lobster pot floats, as were three sperm whales. Of the remaining odontocetes, seven exhibited evidence of the animal's having been entrapped in unidentified nets, two in trawl nets, and one in a longline. In the Icelandic fin whales, it was not possible to identify the form of entanglement gear which had made the scars.

The majority of the entanglements occurred in sea turtles, and there were clear differences among the species. The chelonid turtles (loggerheads, greens, and Kemp's ridleys) were primarily caught in pound nets (44 out of 48 turtles), while leatherbacks were entangled in other types of nets (4 of 10) and in lobster pot lines (6 of 10).

The incidence of death among entangled animals was related to the type of entrapping gear. Those types of gear which can hold an animal underwater were more frequently associated with the animal's death. The odontocetes which showed evidence of net entanglement had all died of drowning. These animals appeared healthy prior to death, exhibiting full stomachs, normal blubber thickness, and no specific disease etiology. One leatherback turtle became entangled in a lobster pot line and could not be freed. This animal also drowned. There was no mortality among the 44 turtles entrapped by pound nets, which only encircle an animal and do not confine it under water.

DISCUSSION

Between the years of 1979 and 1988, 461 stranded and entangled animals were found in the New York Bight. These strandings included 17 species of marine mammals and 4 species of sea turtles, and many of the data were collected from carcasses that had washed up along the shores of Long Island, New York. The prevailing wind and current patterns are such that most carcasses in the Long Island Sound or in the eastern bays are transported to shore, but many of those in the ocean float farther out to sea. Thus, while some areas provide an accurate account, strandings along the entire

ocean shore probably grossly under-represent the number of pelagic animals that are impacted.

The incidence of ingested synthetics varied among species. The observed patterns could be attributed to several ecological characteristics of the animals: feeding behavior, seasonal occurrence, and habitat. The type of synthetic found in 19 of 24 animals was floating or neutrally buoyant plastic. Much of this type of refuse originates on land or comes from recreational boating near shore and concentrates inshore during the summer when human activity is highest. Many of the cetaceans are deep water animals, but during the summer they often move inshore, where they have been observed to be feeding heavily. It is likely that ingestion of synthetic materials increases at this time. Animals that stranded during the winter months, such as seals and most Kemp's ridley turtles, contained no synthetic materials.

The ingestion of synthetics also corresponded to the feeding behavior of animals. The mysticetes and a few odontocetes feed throughout the water column by capturing large quantities of food at a time. Plastics and other floating materials are probably ingested along with prey species. Leatherback turtles feed almost exclusively on jellyfish (Mortimer 1981) and probably actively feed on plastic that resembles their prey. Conversely, the Kemp's ridley feeds very selectively on crabs off the bottom and seals in the New York Bight feed primarily on crabs and benthic fish and neither was found to contain debris. In many cases where synthetics were evident, it was difficult to ascertain the direct cause of death due to the decomposed state of the carcass. However, in some animals, the ingestion of synthetic debris caused serious damage and probably resulted in the death of the animal.

The entanglement data were valuable in determining the effects of different types of debris and fishing gear on the species studied. All of these animals must come to the surface to breathe. Debris in the water column or at the surface, such as floating line, can entangle these animals during their normal activities. Lobster pot float lines proved to be a major source of entanglement for pelagic animals such as fin whales, sperm whales, and leatherback turtles. These lines can be more than 100 m long and virtually undetectable below the surface. Types of active or inactive fishing gear that hold animals below the surface, such as longlines, trawlers, and gillnets, can drown marine mammals and sea turtles. Other types of gear that merely confine animals are not a problem. Most of the Kemp's ridley, loggerhead, and green turtles were caught in pound nets with no observed mortalities.

This study examined the impact of two forms of ocean debris. However, there are many other human activities that can affect marine mammals and sea turtles. Recreational boating contributes heavily to fouling the inland waters, and a large proportion of the animals in this study had been struck by boats. Other problems such as heavy metals, pesticides, and sewage runoff are epidemic in many coastal waters. While their effects on marine life may not be immediate, pollutants may result in health problems and have detrimental effects on the long-term survival of populations.

Martineau et al. (1985) showed that ingestion of toxicants drastically reduced the reproductive rate of beluga whales. It is possible that ingestion of debris and entanglement of animals have similar long-term effects, and the numbers of impacted animals are probably much higher than shown in this study (Kraus 1990)

Although the magnitude of the problems of ocean debris is not yet fully realized, this study indicates that the impact of human activity is not restricted to highly populated areas such as the New York Bight. It occurs globally and is found even in such remote areas as Iceland.

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A REVIEW OF GHOST FISHING BY TRAPS AND GILLNETS

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ABSTRACT

Ghost fishing occurs when lost fishing gear continues to catch and kill animals. This paper reviews what is known about ghost fishing in trap and gillnet fisheries, how the information was obtained and how it has been used, how ghost fishing can be prevented, and what regulatory approaches have been taken to address the problem. Some standard terms are proposed to prevent confusion.

Ghost fishing by traps can occur through several mechanisms. The problem is serious in several fisheries, minor in at least one, and remains unexamined for the majority of trap fisheries. Timed-release devices are simple, inexpensive, and effective at preventing ghost fishing by opening the trap some time after loss. In all Dungeness crab fisheries, such devices are required in crab traps, and other regulations attempt to minimize trap loss. In the American lobster fishery, only Connecticut and Maine address ghost fishing, which is known to be a problem. Ghost fishing by traps is poorly recognized as a problem outside North America.

Ghost fishing by coastal gillnets has been documented in several locations and may persist for several years. For large pelagic gillnets the limited evidence suggests that lost nets form tangled nonfishing masses. More work, both descriptive and experimental, is required to document the nature, extent, and persistence of ghost fishing by gillnets, especially by pelagic gillnets if their use continues.

It is not clear how to prevent ghost fishing by gillnets. Preventive measures suggested to date must be examined for possible side effects.

INTRODUCTION

Ghost fishing can be defined as "the ability of fishing gear to continue fishing after all control of that gear is lost by the fisherman" (Smolowitz 1978a), i.e., when gear is lost, a common occurrence in many fishing operations. The subject was previously reviewed for trap fisheries by Smolowitz (1978a, 1978b, 1978c) and for several gear types by High (1985).

Fishing gear that requires active control, for example trawls, troll gear, and purse seines, may become virtually inert and probably catches insignificant numbers of animals after loss. By contrast, gear which normally fishes passively, such as traps, tangle nets, and gillnets, may continue to fish at significant rates after loss.

This paper looks at what is known about ghost fishing by traps and gillnets, how this knowledge was obtained and used, and what measures can be taken to reduce ghost fishing by traps and gillnets. Other fishing gear may well ghost fish--High (1980, 1987) reports Pacific halibut, *Hippoglossus stenolepis*, striking and being caught by bare longline hooks--but the literature at this stage adds little to High's (1985) review for other gear types.

Why Ghost Fishing May Be a Problem

An increasing proportion of fishing gear is now constructed from nondegradable materials such as stainless steel, other metals, fiberglass, injection-molded plastics, vinyl-coated wire, monofilament netting, and polypropylene twine. Whereas fishing gear made from natural materials deteriorated quickly in the sea--Pacific salmon, *Oncorhynchus* spp., fishing ports all featured tanks of copper sulfate for dipping nets to preserve them--gear made from modern materials lasts much longer in the sea.

The very large volumes of fishing gear now deployed translate to a large volume of lost gear even if the loss rate is small. Some crustacean trap fisheries are so overcapitalized that jurisdictions try to limit the large number of traps used. Hundreds of thousands of kilometers of pelagic gillnets are in use. If this gear ghost fishes when lost, then there is a serious biological and economic problem.

Terminology

Some standard definitions are proposed. First, I use "lost" to describe lost or discarded fishing gear. Previous authors have used "ghost" or "derelict" to describe such gear. However, using "ghost" to mean "lost" creates confusion--the lost gear may or may not actually be ghost fishing. Sutherland et al. (1983) propose a distinction between intact lost gear, still theoretically capable of fishing, and damaged or "derelict" gear that can no longer fish. "Derelict" should be limited to this sense. Where gear loss is simulated experimentally, I use the term "simulated lost" gear.

Second, two types of special openings in traps need careful differentiation. Traps can be modified by openings designed to allow animals to escape (Wilder 1945). These openings have been termed "savings gear" (Jow 1961), "escape vents" (Pecci et al. 1978; Anthony and Caddy 1980), and "escape gaps" (Brown and Caputi 1986). Traps can also be modified by openings or mechanisms designed to release animals from a lost trap. These have been termed "biodegradable sections" (Anthony and Caddy 1980), "timed-release mechanisms" (Blott 1978), "ghost panels" (Krouse pers. commun.), "escape panels" (draft Maine legislation), and "destruct panels" (Hipkins and Beardsley 1970). I suggest that the first type of special opening be called "sublegal escape gaps" and that the second type be called a "timed-release" opening.

GHOST FISHING BY TRAPS

Mechanisms

There is no single mechanism of ghost fishing by traps because traps vary widely in their design, intended mode of capture, target species, and conditions of deployment. To understand ghost fishing it is first necessary to look briefly at trap operation.

Some traps simply attract fish or crustaceans with bait. Although the animals can apparently escape at will, a number are found inside the traps: there is a temporary balance between catch and escape rates. Examples are reef fish traps (Munro et al. 1971; Munro 1974), Australian snapper traps (Dews et al. 1988), and British Columbia prawn traps (Boutillier pers. commun.). Difficult exits in fish traps reduce escapement rates to increase the number of fish in the trap (see Munro 1983).

Escape can be made more difficult by fitting "nonreturn valves" to traps (e.g., Munro 1974). Dungeness crab, *Cancer magister*, traps are fitted with hinged metal gates called "triggers" (High 1976) that permit entry but effectively block exit. Sablefish, *Anoplopoma fimbria*, traps may have similar devices (Hipkins and Beardsley 1970). Homarid lobster, *Homarus americanus* and *H. gammarus*, traps commonly have inner chambers or "parlors" to hinder escape (Pecci et al. 1978; Lovewell et al. 1988).

In the simplest form of ghost fishing, trapped animals die in lost traps and their bodies act as bait (von Brandt 1984). Hipkins and Beardsley (1970, p. 29) state: "It appears then that blackcod (sablefish) pots. . . will continue to fish with dead fish serving as bait to attract new fish which eventually die to attract more fish and so on ad infinitum until the pot deteriorates. . . ." They present indirect evidence for this mechanism. Pecci et al. (1978) suggest this mechanism may operate in lost American lobster traps. For no species has this "autorebaiting" mechanism been conclusively demonstrated.

Traps may be rebaited by species other than the target species. Alaska king crab *Paralithodes camtschatica* traps are rebaited when Pacific halibut or Pacific cod, *Gadus macrocephalus*, enter and die (High and Worlund 1979). Pecci et al. (1978) report a variety of fishes caught and perhaps acting as bait in simulated lost American lobster traps.

Some species of fish are attracted to live conspecifics in an unbaited trap (Munro 1974); for these, ghost fishing might occur without the auto-rebaiting mechanism.

In the simplest model of ghost fishing, trapped animals starve in the traps. Other forms of mortality might be important, causing death sooner. In crustaceans, cannibalism of newly molted individuals may occur. Pecci et al. (1978) observed this in simulated lost American lobster traps; Demory (1971) and Barry (pers. commun.) observed this for Dungeness crabs. Scarratt (1965) reported predation of captured American lobsters by amphipods. Ritchie (1972) and Gabites (pers. commun.) report predation on trapped New Zealand spiny lobsters, *Jasus edwardsii*, by octopus, *Octopus maorum*; Morgan (1974) describes predation by *Octopus* sp. on the Western Australian spiny lobster *Panulirus cygnus*; High (1985) describes attempts by *O. dofleini dofleini* to capture trapped Dungeness crabs. Pecci et al. (1978) reported mortality of American lobsters in simulated lost traps caused by black sea bass, *Centropristis striata*. Trapped crabs may be smothered when the trap is buried by silt (High 1985).

Even when animals manage to escape from ghost fishing traps, they may die as a result of their confinement--High and Worlund (1979) demonstrated this important effect experimentally for Alaska king crabs.

Fishes and crustaceans may enter unbaited traps. This is reported for Hawaiian spiny lobsters, *P. marginatus*, (Paul 1984) in the laboratory; New Zealand spiny lobsters in the field (Gabites pers. commun.); and American lobsters in the field (Pecci et al. 1978; Smolowitz 1978b; but cf. Karnofsky and Price 1989). Dungeness crabs (Breen 1987) and Alaska king crabs (Meyer unpubl. manuscr.) entered empty traps months after simulated trap loss. Munro (1983) describes fish traps that catch fish unbaited. Juvenile reef fishes in Florida use traps as shelter (Sutherland et al. 1983). High and Ellis (1973) found that unbaited traps caught as many reef fish as baited traps. For such traps an autorebaiting mechanism is not necessary for ghost fishing to take place.

In some cases dead crustaceans repel conspecifics. Hancock (1974) demonstrated this effect experimentally for the spiny lobster *P. cygnus*, and also presented evidence that the crabs, *Cancer pagurus*, are not attracted to traps baited with the crab *Carcinus maenas*. Miller (1977) demonstrated in experimental trapping that the Newfoundland snow crab, *Chionoecetes opilio*, are repelled by dead conspecifics, and High (pers. commun.) also reports this for the Alaska king crab. However, Pecci et al. (1978) found that *H. americanus* are not repelled by dead conspecifics. For species repelled by dead conspecifics, the autorebaiting mechanism will not cause ghost fishing.

Thus ghost fishing can occur through a variety of mechanisms: auto-rebaiting, rebaiting by other species, attraction by living conspecifics, or attraction by the trap alone. The trap may kill through starvation or by facilitating cannibalism and predation. For some species, conspecific repellency may prevent or reduce ghost fishing. Ghost fishing may be significant on species other than the target species.

Demonstrations That Traps Ghost Fish

Recovered Lost Gear

Recovery, especially after long periods, of lost gear that contains live and dead animals is good evidence that ghost fishing occurs. Hipkins and Beardsley (1970) recovered nine sablefish traps lost for approximately 6 weeks. These contained dead fish and up to 24 live fish per trap, suggesting that the autorebaiting mechanism was operative.

In Oregon, Demory (1971) retrieved 117 Dungeness crab traps which had been abandoned for at least 6 weeks. They contained 3,629 crabs, 91% of which were legal-sized males. Dahlstrom (unpubl. manusc.) recovered an Oregon Dungeness crab trap, lost for 10 months, containing 20 crabs and 2 empty carapaces. The trap was still in excellent condition. Meyer (unpubl. manusc.) reports that recovered lost Alaska king crab traps "often contain as many as 100 marketable king crab."

Smolowitz (1978a) recovered 18 intact offshore American lobster traps lost for approximately 9 weeks. They contained 24 lobsters weighing a total of 70.8 kg (156 lb). High and Worlund (1979) recovered a snow crab, *Chionoecetes* spp., trap containing 12 king and 14 snow crabs 3 months after loss. Sutherland et al. (1983), using a submersible, found five undamaged fish traps in Florida, lost for an estimated 4-6 months. These held 14 fish, 14 Caribbean spiny lobsters, *Panulirus argus*, and a fish skull.

When lost traps are empty on recovery, it is often inferred that ghost fishing does not occur. For instance, Boutillier (pers. commun.) observed lost prawn, *Pandalus platyceros*, traps from a submersible in British Columbia; none contained prawns and he concluded that ghost fishing did not occur. However, simulated lost Dungeness crab traps that were empty for considerable parts of the year caught and killed crabs (Breen 1987). If traps ghost fish through other than the autorebaiting mechanism, then an empty trap may subsequently kill. Inferences made from empty traps are suspect unless made over large numbers of traps and over several seasons.

Another inference is often made from the way catch rates fall as soak time increases. Traps left to soak for too long give poor catches; the inference is that most of the catch escapes after the bait ceases to attract. Then by extension ghost fishing is inferred not to be a problem. Examples include the Tasmanian spiny lobster, *J. novaehollandiae* (Kennedy pers. commun.), British crabs and lobsters (Bannister pers. commun.), and Dungeness crabs. However, in Dungeness crab traps the catch rate declines with increased soak time, yet lost traps continue to catch and kill at a slow rate (Breen 1987). So ghost fishing may occur in the long term despite apparent short-term escapement.

Trap Loading Experiments

Ideally, all ideas about ghost fishing should be tested experimentally. Three approaches have been used: experiments in which traps are loaded and escape rates or mortality rates measured, laboratory

observations that simulate fishing, and field experiments with simulated lost traps.

Munro (1974) found that 50% of reef fishes escaped from Antillean fish traps after 14 days; this implies a 5% escapement per day and 23% retention after a month. These rates suggest that ghost fishing is likely to occur in such traps. However, Munro (1983) estimated an escapement rate of 12% per day from "Z" fish traps, a rate implying only a 2% retention after 30 days.

Sheldon and Dow (1975) loaded 98 tagged American lobsters into 35 unbaited simulated lost traps and checked the traps by diving and hauling for nearly 2 years. The traps continued to catch lobsters, of which 12-18% died in the traps, demonstrating for the first time ghost fishing for American lobsters.

Newfoundland snow crab traps were loaded and examined at intervals by diving (Miller 1977). After 3 weeks no crabs had escaped. Miller then tested the mechanism of ghost fishing in this species by fishing with four treatments. Unbaited traps and traps baited with dead crabs caught nothing; on average squid-baited traps caught 31 crabs per trap; traps baited with a mixture of dead crabs and squid caught 7 crabs per trap. Miller concluded that dead snow crabs repel conspecifics and that the only loss from ghost fishing would involve those crabs originally attracted by the bait.

High and Worlund (1979) observed a 20% retention rate for legal size Alaska king crabs and 8% for sublegal crabs in experimental traps after 18 days. Mortality in standard traps was 2-7% over this period.

Muir et al. (1984) baited Dungeness crab traps daily and observed that 35% of the captured crabs died in the traps. High (1985) placed Dungeness crabs in traps with and without triggers and sublegal escape gaps. The mortality in traps with functional triggers and sublegal escape gaps was 17% after 12 days, confirming ghost fishing as a problem with these traps.

Laboratory Observations

Paul (1984) observed that Hawaiian spiny lobsters in a large tank normally did not escape from traps. The trap lids "had to be removed to prevent them from becoming permanently trapped inside."

Behavior of reef fishes around traps was observed in a large tank by Harper and McClelland (1983, cited by Heneman and Center for Environmental Education (CEE) 1988). Most species appeared to learn to escape, leading to "an equilibrium state. . .with frequent movements in and out of the trap."

Booth (pers. commun.) used a time-lapse camera in a large tank to record the behavior of *J. edwardsii* around simple cane traps, as used in the New Zealand fishery, and parlor-type traps not used in the fishery. There was a rapid turnover of lobsters in the simple trap, but greatly

reduced escapement in the parlor traps. Booth concluded that ghost fishing is probably not a problem for the cane traps, but could be a problem if more complex traps were introduced. Plastic truncated-cone entrances on the top of the trap appear to limit escape in this species in large laboratory tanks (Breen unpubl. data; Gabites pers. commun.).

Field Experiments

Information from trap loading and laboratory studies must be treated with caution: problems with extrapolation from the laboratory to the field and from short to long term must be carefully considered. Possibly the best information comes from underwater observations of simulated lost traps. Tagging of trapped individuals by divers can be used to follow turnover.

Pecci et al. (1978) reported only 30% escapement in American lobsters entering simulated lost traps observed by divers. Mortality rate was 25%. The authors estimated that a ghost fishing trap caught at a rate near 10% that of a surface-hauled trap, confirming ghost fishing as a problem in this fishery.

Breen (1987) simulated 10 lost Dungeness crab traps in a sheltered bay for 1 year, during which approximately 100 crabs died in the traps. At the end of the study, traps were still killing crabs at a steady rate. The results cannot be generalized directly to other Dungeness crab fisheries. For instance, many traps lost off high-energy beaches are destroyed or put ashore by wave action.

Western Australian snapper, *Chrysophrys auratus*, traps were observed in the field with underwater video (Anonymous 1984; Dews et al. 1988; Moran and Jenke 1989) partly to examine possible ghost fishing (Bowen 1961). Fish seemed capable of leaving traps easily and some even swam out "in reverse." Moran and Jenke (1989) simulated lost traps for various periods from 1 to 21 days. Catches were similar to commercial catches after 15 min, indicating that cumulative catching did not occur. These workers concluded that ghost fishing is not a problem with snapper traps. However, three fish were dead in the 21-day trap, suggesting that some ghost fishing may take place.

Hawaiian spiny lobsters appear to move out of simulated lost traps once the bait has deteriorated (Okamoto pers. commun.; Parrish pers. commun.).

Rates of Trap Loss

Traps are lost for many reasons. Simple vessel traffic and towboating may sever buoy lines or drag traps into water deeper than the buoy line. Weak or chafed buoy lines may break. Buoys may become detached from the buoy line, or may be attacked by marine birds (Smolowitz 1978b) or mammals (High 1985). Storms or strong currents may "drown" traps either directly or by rolling them over the bottom, wrapping the buoy line around the trap

(Smolowitz 1978b; Sutherland et al. 1983; von Brandt 1984). Traps set on rocky ground may snag and be unrecoverable (Bowen 1961).

Traps may be carried into deep water, or buoy lines cut, by other fishing activities such as trolling, trawling and gillnetting. When traps are set on ground lines, ground lines may be intentionally cut when lines become fouled. Internecine buoy line cutting or ground line cutting may result from unresolved fishing disputes (Smolowitz 1978b; Breen 1987). In some areas vandals cut buoy lines (Sutherland et al. 1983).

Estimates of trap loss rate must be obtained through surveys or industry interviews. These give the total loss of traps, which might include stolen traps.

American Lobster Traps

For the U.S. American lobster fishery, Smolowitz (1978a) cites anecdotal estimates of the annual loss of traps as 20-30% along the Atlantic seaboard. In the offshore lobster fishery he suggests that 40,000 all-metal traps may have been lost during the period 1971-78. In the inshore fishery Krouse (pers. commun.) suggests an annual loss rate of 5-10%. Based on a 1987 estimate of 1.87 million traps fished, this leads to an annual loss estimate of 93,500-187,000 traps lost annually. An unpublished study (CEE 1987) cited by Heneman and CEE (1988) estimated an annual loss of 500,000 traps annually. In Rhode Island a logbook study led to an estimate of 10-15% annual loss (Fogarty pers. commun.).

In Newfoundland, no estimates have been made of lobster trap loss rate, but divers observe few lost traps on the fishing grounds. Many lost traps are washed ashore (Ennis pers. commun.). Losses have not been estimated in the rest of the Canadian lobster fishery.

Dungeness Crab Traps

In California, 100,000 Dungeness crab traps are estimated lost each year (Kennedy 1986). Some silt into the bottom, but others could fish for an estimated 2 years. In Washington State, Northup (1978, cited in Muir et al. 1984), estimated that 17.6% of the coastal Washington State crab traps were lost in 1975-76, considered a typical year. Barry (pers. commun.) estimated mean annual loss in the same fishery as 11.9%. He considers that ghost fishing traps are <50% of the total loss and may be as low as 10%. In the Puget Sound portion of the fishery, gear loss was estimated from a questionnaire survey to be 15% (Bumgarner pers. commun.). Breen (1987) estimated Fraser River Dungeness crab trap loss as 11%, based on a questionnaire survey. About half those surveyed thought that half the lost traps were ghost fishing.

Thus in several coastal trap fisheries, annual trap loss rates are on the order of 10-20%. American lobster and Dungeness crab fisheries are both cases where more traps than optimum are fished (and thus lost) (Bell and Fullenbaum 1986; Methot 1986). Cumulative trap losses are a cause for concern in fisheries where ghost fishing is known to occur.

The Fate of Lost Traps

Not all lost traps become ghost fishing traps even where ghost fishing is a problem. Smolowitz (1978b) reviews sources of trap destruction. Storms destroy or strand many inshore American lobster and Dungeness crab traps in exposed locales. Burial by storm action or alluvia occurs quickly in some Dungeness crab fishing areas (Hipkins 1972, cited in Smolowitz 1978b; Breen 1987).

Untreated wooden traps are destroyed by borers in a relatively short time, but treated wooden traps may last up to 2 years (Smolowitz 1978b; Fogarty pers. commun.). Twelve percent of the wooden traps used by Sheldon and Dow (1975) were so damaged by lobster chelipeds that escape became possible. Increasingly, however, traps are made from metal (Acheson 1982) or synthetic materials. Averill (pers. commun.) believes that "wooden" American lobster traps last as long as wire traps when lost. Long-term experiments are required to determine the fishing lifespan of various trap types.

High and Worlund (1979) estimate that metal-framed, synthetic mesh-covered Alaska king crab traps could have an effective longevity of 15 years after loss. Breen (1987) found that metal-framed, stainless steel-covered Dungeness crab traps were in excellent condition after a year's submersion. Electrolytic corrosion probably destroys most metal traps eventually. New designs include plastic traps (e.g., Piatt 1988) and vinyl-coated mesh (e.g., Maynard and Branch 1988), which might last for decades. The present Maine trap inventory is 50-60% vinyl-coated wire (Averill pers. commun.).

Note that much of the information just presented is based on short-term studies. The real fate of lost fishing gear has not been well studied.

Impact of Trap Ghost Fishing

How much fishing takes place by ghost-fishing traps? To answer this for a specific fishery requires 1) estimates of the number of traps fished and the loss rate, 2) an assumption about the percentage of lost traps that ghost fish, 3) an estimate of the rate of mortality in ghost-fishing traps, and 4) an estimate of the effective ghost fishing lifespan of a trap. Ideally, for requirement 3 one should also know the natural mortality rate, because some individuals killed by ghost fishing would have died before commercial capture. Many individuals would also have grown before commercial capture. However, the unavoidable imprecision of the other estimates implies that only a crude answer can be obtained in any case.

For the Newfoundland snow crab fishery, Miller (1977) used spot interviews to estimate trap loss at 8%, and combined this with commercial catch rates and experimental observations to obtain an estimate of ghost-fished catch of 10 metric tons (MT) annually. Smolowitz (1978b) estimated the impact of ghost fishing in the U.S. portion of the American lobster fishery. The estimated annual ghost fishing catch was 670 MT, worth an

estimated 1978 US\$2.5 million. From a 1976 study using different assumptions (CEE 1987, cited by Heneman and CEE 1988), the economic loss of just those lobsters within traps at the time of loss was estimated at 1976 US\$2.5 million. Krouse (pers. commun.) assumed a loss rate of 5% in the U.S. American lobster fishery and that traps last for 2 years and take two lobsters per year. This leads to an estimate of 204 MT lost to ghost fishing annually, worth 1989 US\$1.2 million. This is a conservative estimate, because it is based on the low end of the range of trap loss estimates.

Breen (1987) estimated the impact of ghost fishing in one part of the British Columbia Dungeness crab fishery, using loss rates and lifespan estimates from an industry survey and experimental ghost fishing data. He estimated the ghost-fished catch to be 7% of reported landings, worth about 1985 Can\$80,000.

For the sablefish fishery of British Columbia, Scarsbrooke et al. (1988) used trap loss rate from an industry survey, the commercial catch rate, and simple assumptions about turnover rate, trap lifespan, and timed-release device effectiveness. For traps lost from 1977 to 1983, before timed-release devices were fully employed, the estimate of ghost fishing catch was approximately 300 MT annually, compared with landings of 1,000-4,000 MT.

These cases illustrate that ghost fishing can be substantial. I can find no fishery for which the impact of ghost fishing on stocks has been determined, or where ghost fishing is addressed by stock assessments or management plans. In Oregon, where traps are required to incorporate timed-release mechanisms, biologists consider that ghost fishing, although subtracting from the potential catch, would have no stock-recruitment effect. The size limit is set so that all legal-sized males could theoretically be taken without affecting reproduction (Demory pers. commun.).

Prevention of Trap Ghost Fishing

Remedial measures may either reduce trap loss or prevent lost traps from killing. A simple way to reduce trap loss is to reduce the number of traps fished (Smolowitz 1978b). Effort is excessive in many fisheries, so this approach is often desirable for that reason alone. The extreme solution, vessel trap limits or transferable trap entitlements, is extremely expensive to enforce and therefore was rejected as a management option in the New Zealand *J. edwardsii* fishery (Anonymous 1987).

Trap designs can be improved to reduce storm and current losses caused by traps rolling on the bottom (see Smolowitz 1978b). Losses caused by vessels can be reduced by prohibiting buoyed traps in areas of heavy traffic. In Washington State, trap-free lanes for towboats have been established to minimize trap loss from that source (Bumgarner pers. commun.). The Washington Department of Fisheries also facilitates coordination between trap and net vessels to avoid gear collisions. In the Canadian sablefish fishery, ground lines must be buoyed at each end. In practice, the marking employed far exceeds the minimum standard required (McFarlane pers. commun.). In the Puget Sound recreational trap fisheries

of Washington State, regulations require solid buoys (to prevent losses from puncture) and nonfloating buoy lines (to prevent loss from vessel traffic) (Bumgarner pers. commun.).

The large literature on sublegal escape gaps shows that they greatly reduce catches of sublegal crustaceans, presumably through escapement (e.g., Cleaver 1949; Fogarty and Borden 1980; Brown and Caputi 1986; see review in Smolowitz 1978c). Because escape gaps reduce trap saturation effects (Miller 1979), they may lead to increased catches of legal animals.

Ghost fishing mortality was reduced for sublegal American lobsters by sublegal escape gaps in simulated lost traps (Pecci et al. 1978; Smolowitz 1978a). High (1985) found greatly increased sublegal escapement in simulated lost Dungeness crab traps fitted with sublegal escape gaps. Breen (1987) found that as many sublegal as legal Dungeness crabs died in simulated lost traps fitted with appropriate sublegal escape gaps, but the absolute catch rates of legal and sublegal crabs were unknown. Sublegal crabs may have had a high turnover rate in the traps.

Measures to prevent lost traps from ghost fishing usually involve some deliberate failure (timed-release) in a trap component to open the trap or create a new opening for escapement.

Natural fiber twine can be used either to make a timed-release panel or to sew a timed-release panel shut. Panels can also be made from untreated softwood. Blott (1978) tested a variety of materials with potential for use as timed-release elements in traps. Jute and manila twine and steel wire appeared to be realistic, while wool and leather were not.

In Maine, various materials have been tested for use in closing timed-release openings (Averill pers. commun.). Industry was given traps with many openings secured with test materials and asked to fish them during their regular season. Mild steel hog rings appear to last the desired time (ca. 200 days), and are consistent in their total degradation time. Cotton twine and sisal twine are also good candidates for this purpose.

Scarsbrooke et al. (1988) tested failure rates of several binding materials for timed-release openings in sablefish traps. They also fished traps with three types of opening in alternation with control traps to measure the effectiveness of timed-release openings. Triangular or square openings were more than 90% effective in allowing trapped fish to escape; simple "slashes" were less effective. They concluded that appropriately shaped timed-release openings eliminated the problem of ghost fishing in these traps.

Plastic crab and lobster traps in Florida (Piatt 1988) have a rectangular opening which the user fills with a timed-release device such as a plywood panel.

Blott (1978) describes a solid timed-release panel made from galvanized steel and held shut with natural twine or a degradable metal ring. The panel can also incorporate the sublegal escape gap, leading to the name

"catch escape panel." Blott tested various materials for suitability as catch escape panels; galvanized sheet steel seemed most appropriate. Pecci et al. (1978) tested such panels in simulated lost American lobster traps and concluded that such panels "are an effective means of releasing entrapped lobsters." Traps with this type of panel are now commercially available from a Maine manufacturer (Lazarus 1988). However, Averill (pers. commun.) considers that the combination of sublegal escape gaps and a timed-release opening leads to confusion of two separate management issues.

In California, magnesium pins are used to hold together the two halves of plastic or fiberglass traps or to attach the lids of plastic and fiberglass traps (Estrella pers. commun.).

Dungeness crab traps are serviced through the "lid," a hinged section of the top secured by a hook attached by a rubber strap from the side of the trap (High 1976). A timed-release hook, or hook attachment, would allow the trap to open. Breen (1987) unhooked 10 simulated lost traps that had ghost fished for a year. Over a week, 22 of 29 trapped crabs escaped and no new captures were observed. Thus a timed-release device that unhooked the lid would probably be effective in this type of trap.

It is possible to make plastics that are degraded by organisms, light, oxidation, other chemical reaction, and dissolution (see review by Andrady 1988). Various degradable plastic compounds designed specifically for the fishing industry are now being tested (Gonsalves et al. 1989, Gonsalves 1990). Japanese chemists are designing "bacterial co-polymers" which degrade slowly into natural chemicals in water (Doi et al. 1988).

Premature failure of timed-release elements reduces industry acceptance of the concept (Smolowitz 1978b). The early failure of a batch of hog rings used to close timed-failure panels in lobster traps resulted in industry resistance to the devices in Maine (Anonymous 1988; Averill pers. commun.). A similar experience in California led to delayed legislation (Estrella pers. commun.). Material failure rates vary widely with local conditions and probably cannot be predicted accurately. Agencies proposing timed-release regulations must conduct widespread materials testing to find a mechanism that will both fail reliably after the desired time and not fail prematurely. Studies conducted by the industry under actual fishing conditions are more likely to be accepted by the industry.

The dollar and time costs of timed-release modifications are important to acceptance by industry (High and Worlund 1979). Breen (1987) calculated the annual economic cost of Dungeness crab trap ghost fishing done in 1985 as Can\$1.46 per trap in use, and suggested that annual modifications must therefore cost less than this. This simple study appears to be the only published cost-benefit analysis of the problem. Other managers consider that "off-the-cuff cost-benefit analysis would indicate that [ghost fishing] should be addressed" (Averill pers. commun.).

Finally, Smolowitz (1978b) suggests development of "habipots" that catch animals seeking them as shelter. Such traps would not entrap animals

and thus would have only biologically positive effects when lost. Some *Octopus* traps operate on this principle (Mottet 1975).

Regulations to Prevent Trap Ghost Fishing

The American lobster and Dungeness crab fisheries are interesting to examine for regulations designed to minimize ghost fishing. In both fisheries ghost fishing is known to occur, trap losses are high, and the fisheries take place over several jurisdictions in two countries with differing management approaches.

Dungeness Crabs

California requires all traps to incorporate timed-release devices or openings. These may be trap lid hooks made of soft steel <6 mm diameter, lid hooks attached to the strap with single loops of natural fiber twine, any modification of the upper mesh secured with natural fiber to create a 125-mm diameter hole, or magnesium pins as discussed above. Testing of these materials has been carried out, and cotton twine is the preferred option (Estrella pers. commun.). All traps or ground lines of traps must be buoyed and the buoys marked with identification markings.

Oregon requires Dungeness crab traps to contain a timed-release device as in California (Demory pers. commun.). Individual traps must be buoyed and marked.

Since October 1988, Washington also requires timed-release devices as above but not including the mild steel hook; openings must be unimpeded, at least 76 x 127 mm and closed with natural fiber. Washington also has buoy and buoy line standards described earlier.

In British Columbia, Fisheries and Oceans Canada introduced a regulation in 1990 requiring a single loop of specified cotton twine in the lid strap and nonfloating buoy lines. Traps or ground lines must be buoyed with marked floats, but this regulation is often ignored (Breen 1987).

In Alaska, Dungeness crab traps are required to have timed-release devices (Koeneman pers. commun.). At least as early as 1974, Alaska sablefish traps were required to incorporate timed-release panels (Hipkins 1974, cited in High and Worlund 1979).

Alaska also requires that "traps left unattended for over 2 weeks must have bait removed and doors secured open as protection against ghost fishing." This is the only regulation dealing with ghost fishing listed by Miller's (1976) review of crab management regulations in North America, demonstrating the relatively recent recognition of the problem.

Most other major trap fisheries on the Pacific coast have similar regulations. Scarsbrooke et al. (1988) describe the requirement for a timed-release panel in the sablefish fishery. In this case the fishing industry actually included such devices before being regulated. Regulations governing a new trap fishery for hagfish, *Eptatretus* spp., require timed-failure openings in British Columbia and Oregon (Harbo pers. commun.).

American Lobsters

In the United States, Connecticut has been the only jurisdiction to require incorporation of a timed-release panel into the trap. Maine drafted legislation in 1982, which will take effect in 1990 (Krouse 1989), requiring a timed-release panel at least 95 mm square, made of untreated natural material: twine <5 mm diameter, ferrous metal less than about 2.5 mm diameter, or softwood. In the federally controlled part of the fishery, degradable fasteners closing a timed-release opening will be required in 1992 (Fogarty pers. commun.).

In the federally regulated portion of the fishery, lobster traps must be marked with the owner's identification number, and traps set on ground lines must be marked with a buoy or flagpoles and radar reflectors, depending on how many traps are set.

In the Canadian fishery, no regulations are directed at ghost fishing. Anthony and Caddy (1980) recognized the problem and recommended that timed-release panels or "links" be included in all traps and especially deepwater traps.

GHOST FISHING BY GILLNETS

Mechanisms

Gillnets work by trapping animals in the mesh of the net; ghost fishing is a simple continuation of the gillnetting process after the net is lost.

A wide variety of species are targeted with many types of gillnet worldwide (see Uchida 1985 for a comprehensive review). In comparison with the trap fisheries reviewed above, there has been little work on ghost fishing by gillnets. This may reflect failure to recognize a problem: Herrick and Hanan (1988) review problems caused, inter alia, by California gillnets without considering ghost fishing.

Pelagic or drift gillnets are used by Japan and Taiwan in the North Pacific to catch salmon and squid (Uchida 1985), and in the South Pacific by Japan, Korea, and Taiwan to catch albacore and skipjack tuna (Hinds 1984; Murray 1988). Ghost fishing in pelagic gillnets may be overshadowed by their incidental catch performance. They catch a long list of other nontarget species including fishes, birds, turtles, and marine mammals. Even reindeer have been reported caught by gillnets (Beach et al. 1976). Sloan (1984) and McKinnell et al. (1989) give extensive species lists in the incidental catch in Japanese squid gillnetting off British Columbia. In the same fishery Jamieson and Heritage (1987) estimate the catch of birds at one per 18 km of net set, the catch of mammals at one per 140 km. Harwood and Hembree (1987) estimate the incidental catch of cetaceans in pelagic gillnetting off northern Australia, 1981-85, to have been on the order of 14,000 individuals. Incidental catches of cetaceans are also a serious problem in coastal gillnet fisheries. Read and Gaskin (1988) estimated the catch of harbor porpoises, *Phocoena phocoena*, by groundfish

gillnets in the Bay of Fundy, concluding that the incidental catches threaten the population. Recreational gillnetting is a major threat to the endangered Hector's dolphin, *Cephalorhynchus hectori*, in New Zealand (Dawson 1990).

Demonstrations of Gillnet Ghost Fishing

Recovered Lost Gear

In Iceland, synthetic cod gillnets were found a "fairly long time" after loss (von Brandt 1984); they appeared to be fishing actively based on the number and appearance of fish.

Way (1977) described catches of live fishes and crabs in lost demersal Newfoundland cod gillnets retrieved with purpose-designed dragging gear. He concluded that lost gillnets continued to fish "at a declining rate."

DeGange and Newby (1980) described finding a drifting 3.5-km pelagic gillnet lost for at least a month. The net contained 99 birds and 78 fishes. Live birds appeared to be attracted to the net, perhaps by the material already caught, and many of the fish were fresh. These authors confirm the fears of Bourne (1977) that lost gillnet fragments continue to catch and kill birds.

High (pers. commun.) found a lost salmon gillnet with fish skeletons, diving ducks, and seals, *Phoca vitulina*, in varying states of decay, indicating the net continued to kill these animals.

Underwater Observations

After discovering lost salmon gillnets in Washington, High (1985) used scuba to observe them for 6 years. The nets continued to catch crabs, fishes, and birds for 3 years. One net 180 m long contained an estimated 1,000 female crabs (High pers. commun.).

In New England, Carr et al. (1985) made observations from a submersible. They describe fishes entangled in nets estimated to have been lost for at least 2 years. Observations were continued for 3 years from a submersible and remotely operated vehicle (Carr and Cooper 1987; Carr 1988). Nets lost for 3-7 years continued to catch a variety of species, including spiny dogfish, *Squalus acanthius*; American lobsters; and bluefish, *Pomatomus saltatrix*. Later observations on one net indicated that gadoid fish successfully avoided the net, but crabs, *Cancer irroratus* and *C. borealis*, continued to be killed. Carr and Cooper (1987) estimated that lost nets were fishing at approximately 15% of the rate of commercial nets.

Dennis Chalmers (pers. commun.) reported finding a British Columbia herring (*Clupea harengus pallasii*) gillnet lost for at least 4 years: "This net was all bunched and tangled up against a rock ledge in 15 ft [4.6 m] of water and, at the time, there were a few rockfish [*Sebastes* spp.] trapped inside it." Another net found in 11-12 m depth had been lost for at least

7 years. It had no cork line, but the net had enough buoyancy to sit in fishing position and contained several fresh herring.

As in crustaceans, decaying fishes of some species may repel conspecifics. It is believed in New Zealand, for instance, that dead rig, *Mustelus lenticulatus*, and rig offal near a net reduce net catches (Bradstock pers. commun.). This effect might reduce ghost fishing for some species, but no formal research appears to have been conducted.

Schrey and Vauk (1987) reported that more than 2.6% of gannets, *Sula bassana*, visiting Helgoland become entangled in lost gillnets, which caused 30% of the total gannet mortality observed.

Field Experiments

Two simulated lost demersal gillnets were observed by divers in New England (Carr et al. 1985). The nets continued to catch fishes and crabs over 2 1/2 months of observation.

Kim Walshe (pers. commun.) observed simulated lost inshore gillnets by diving for a year in New Zealand. The nets were partly disabled by algal growth and wrapping up, but continued to catch and kill some fish at intervals through the year. Rock lobsters, *J. edwardsii*, are attracted to the fish and are themselves caught by lost inshore gillnets (Anonymous 1978).

Rate of Gillnet Loss

Storms can break gillnets or break off the end markers. Vessels and trawls may run over or cut gillnets. Marine mammals and large fishes may break and carry away nets. In northern waters ice causes gillnet loss (Way 1977). Way also suspected that some nets were simply abandoned at the end of the season. Net fragments may simply be discarded (Gerrodette et al. 1987). In inshore gillnet fisheries, nets snag on obstructions and are lost.

In the New England groundfish gillnet fishery, loss of nets was investigated by CEE (1987, cited by Heneman and CEE 1988). The study examined claims for lost gear made under a U.S. Federal act providing for compensation for gear loss caused by foreign fishing activities. For 1985 and 1986, claims were made for 48 and 29 km of net, respectively. It is unknown what proportion of the total net loss this represented.

Fosnaes (1975) estimated that 5,000 Newfoundland cod, *Gadus morhua*, gillnets were lost annually. Way (1977) conducted a program of lost net retrieval on commercial grounds, finding 148 nets in 48.3 h in 1975 and 167 nets in 53.5 h in 1976.

The density of lost demersal gillnets on a commercial ground in New England was estimated from a submersible by Carr et al. (1985). They found 10 lost nets over 40.5 ha of bottom in 37.5 h search time.

For large pelagic gillnets, a major concern is the tremendous quantity of net in the water. Eisenbud (unpubl. manusc.) estimated that 5,000 km

of net were used in the Japanese North Pacific salmon net fishery alone. Uchida (1985) estimated that 170,000 km of pelagic net were used in 1984 in the North Pacific. Coe (1986) estimated that more than 1.6 million km of squid net were used by Japan, Republic of Korea, and Taiwan in 1985. Even a very small loss rate results in a very large estimate of lost net.

Pelagic gillnets are lost from most of the same causes as coastal gillnets. Because of their great length (12-15 km), these nets are vulnerable to vessel traffic. In the Japanese fishery, intact nets are easier to recover than fragments because radio buoys and lights are installed at each end; most nets recovered by Japanese observers were fragments (Morimoto pers. commun.). Additional causes of loss suggested by Eisenbud (unpubl. manusc.) are desertion of nets in prohibited areas after removal of end markers, and simple discard of old netting. A fisheries observer, Goldblatt (1989), describes a pelagic gillnet vessel entangling her own net in the propeller, then cutting away and discarding a large fragment.

Eisenbud (unpubl. manusc.) reported an estimate that 0.06% of Japanese salmon pelagic gillnet is lost at each set. Gerrodette et al. (1987) report an estimate of 0.05%. They consider this estimate to be low, but Morimoto (pers. commun.) considers that the loss rate would be lower in the squid gillnet fishery because of calmer sea conditions. Tsunoda (1989) observed a Japanese pelagic squid gillnet vessel for 4 weeks and observed no gear loss. When nets were severed by vessels, Tsunoda reports that the crew quickly recovered the subsections. Eisenbud (unpubl. manusc.) estimated annual loss from the Japanese North Pacific squid and salmon gillnet fisheries to be approximately 2,500 km of net.

The density of lost gillnet material can be estimated at sea from transect surveys (Baba et al. 1990; Day et al. 1990). However, the absolute density of lost nets is very low, net fragments cannot be seen from a significant distance, and the tendency is for drifting debris to become nonrandomly distributed by winds and currents. Assessing the impact of lost gillnets through direct surveys is therefore difficult.

Fate of Lost Gillnets

Gillnets are usually made from synthetic materials which can last for long periods of time. High (1985) observed that lost salmon gillnets continue to kill birds and fish for 3 years, and estimated that crabs may be killed for at least 6 years. The direct observations of Chalmers (pers. commun., described above) on herring gillnets tend to support these estimates.

In inshore waters, algal growth on sunken nets may stop fishing by making the nets highly visible to fishes and birds (High 1985; Dennis Chalmers pers. commun.), but Kim Walshe (pers. commun.) reports that fish are caught even in overgrown nets. Strong currents cause the net to tangle lead line over cork line (Way 1977) or end over end (High 1985). High suggests that rolled netting stops catching birds and fishes but may continue to catch crabs. Drift macrophytes and the catch of fish and crabs may cause the net to sink and stop fishing efficiently (Way 1977; Carr et

al. 1985; Millner 1985, cited in Heneman and CEE 1988). Dogfish caused twisting of the demersal gillnets observed by Carr et al. (1985). These authors found three main types of lost net configuration, and speculated that these related to how the nets were lost.

Gerrodette et al. (1987) attached radio transmitters to four sections of gillnet 50-1,000 m long, then monitored the simulated lost nets. The shortest net "collapsed" very quickly, but the largest net remained in fishing condition for at least 10 days. The authors estimated that a 1-km net would remain in a fishing configuration for several weeks.

In a similar study, Mio et al. (1990) examined five simulated lost pelagic gillnets, each 1,200 km long, for nearly 4 months. At the end of this time all nets had twisted themselves together end for end to form a large mass. One net completed this process in 20 days; the others took longer.

The wrapping up of nets may be accelerated by storms. Sloan (1984) observed that squid gillnets off British Columbia became tangled at wind speeds >65 km/h.

Merrell (1984) estimated that netting at sea survives for <10 years. This estimate is based on "aging" nets found stranded.

Prevention of Gillnet Ghost Fishing

As with traps, the most effective way to prevent ghost fishing is to prevent gear loss. In the Japanese pelagic gillnet fishery, vessels are required to mark nets with a radio buoy at one end and radar reflectors at both ends. Radio communication is used to deflect vessel traffic around the nets. Discarding of netting is prohibited, and old netting is disposed of on land (Morishita pers. commun.).

Gillnets could be hung from the cork line with natural fiber twine (Way 1977; von Brandt 1984). In theory when the net is lost the twine would rot, and the lead line would pull the net into deep water. This idea is being examined experimentally for coastal gillnets in New England (McKenzie pers. commun.). The tendency of nets to become tangled (lead line over cork line) might prevent sinking, but would also reduce ghost fishing potential. There is also a danger that sinking the net simply transfers a surface ghost fishing problem to the bottom, as suggested by the salmon gillnet observations.

In British Columbia, a proposal to require herring gillnets to be hung with cotton twine has been drafted, but is still under discussion with industry (Dennis Chalmers pers. commun.).

I am aware of no research into degradable materials for use in the web itself. The use of natural fiber for gillnets would be a backward step because of the massive effort required to maintain and preserve nets during fishing. Gillnets are commonly made from monofilament nylon (Uchida 1985), whereas the major effort in degradable plastics has been aimed at poly-

ethylene or polyolephanes (Scott 1990) or composites of polyethylene or polypropylene and natural material (Blott pers. commun.). A potential problem is that degradable nets would form many smaller net fragments instead of one large one.

CONCLUSIONS

Ghost fishing has not been well studied. Significant information exists for only two gear types: traps and gillnets. The importance of ghost fishing as a potential problem is underscored by very large volumes of fishing gear in use, high gear loss rates in many fisheries, and the widespread use of nondegradable materials such as plastics and stainless steel for fishing gear construction.

The fishing behavior of lost traps has been examined for only a handful of fisheries, mostly in North America. For most of the world's many trap fisheries, the impact of lost gear has simply not been addressed.

Ghost fishing by traps can operate through several mechanisms depending on trap type and the target species. Where impact has been estimated, ghost fishing sometimes emerges as only a small problem (e.g., Newfoundland snow crab and Western Australian snapper); in other cases (American lobsters, Dungeness crabs), ghost fishing is clearly an important biological and economic waste.

Modifications to stop traps from ghost fishing are simple and effective, and can be inexpensive. Such modifications are quick and easy to service once installed. Management agencies should determine whether ghost fishing is a problem in specific trap fisheries. If it is, they should conduct research into material failure rates and require timed-release devices or panels in all traps. Appropriate and properly designed research is required both to convince the industry of the problem and to develop effective timed-failure devices for specific situations.

For Dungeness crab fisheries, all jurisdictions now recognize the ghost fishing problem and attempt to control it. In the American lobster fishery, where ghost fishing was well documented much earlier, most jurisdictions have still not addressed the problem.

In the American lobster and British Columbia Dungeness crab fisheries, the amount of waste caused by ghost fishing would not have been recognized without appropriate experimentation. In no fishery should ghost fishing be rejected as a serious potential problem until proper research has been conducted.

Ghost fishing has been documented in a variety of coastal gillnet fisheries. Lost nets may kill fishes, crabs, birds, and seals for several years. Loss rates of coastal gillnets have not been estimated, but at least two studies indicate a substantial density of lost demersal gillnets on commercial fishing grounds.

The situation in pelagic gillnets is less clear. Loss rates are poorly estimated. At least one study indicates that ghost fishing and

continuing entanglement of birds occurs; other studies suggest that pelagic nets form tangled nonfishing masses in a short time. Further information is needed in two areas: documentation of lost gear encountered at sea, and direct study of the fishing behavior of lost pelagic gillnets.

Short of preventing net loss or prohibiting gillnetting, it is not clear how to prevent ghost fishing in gillnets. Studies of preventive measures such as using degradable hangings are embryonic. Preventive measures may simply change the form of the problem. Side effects of intended preventive measures must therefore be examined carefully.

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AN EXPERIMENTAL STUDY OF DERELICT GILLNET
FRAGMENTS IN THE CENTRAL PACIFIC OCEAN

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ABSTRACT

An experiment designed to investigate the behavior and fate of derelict gillnet fragments was initiated in August 1986 in the central Pacific Ocean. Four fragments of high-seas squid gillnet, varying in length from 50 to 1,000 m, were observed closely for 3 days and subsequently tracked for up to 10 months by satellite. The net fragments changed length, shape, heading, and location under the influence of wind and current. The time a net remained open in a fishing configuration varied from hours to weeks, depending on its initial length. The nets drifted at an average speed of 15 km/day, but with frequent changes in direction, they remained in the general vicinity of the Hawaiian Archipelago. The complex movement of the net fragments means that predicting the drift of marine debris is an oceanographic problem that requires detailed knowledge of surface currents and wind.

INTRODUCTION

The amount of debris in the world ocean is a matter of increasing concern, both to the scientific community and the public at large. The Workshop on the Fate and Impact of Marine Debris (Shomura and Yoshida 1985) focused attention on the problem, and the National Marine Fisheries Service (NMFS) shortly afterwards established a program (Coe and Bunn 1987) to coordinate research, public awareness, and mitigation efforts.

In R. S. Shomura and M. L. Godfrey (editors), Proceedings of the Second International Conference on Marine Debris, 2-7 April 1989, Honolulu, Hawaii. U.S. Dep. Commer., NOAA Tech. Memo. NMFS, NOAA-TM-NMFS-SWFC-154. 1990.

One source of marine debris is fishing operations. The amount of fishing gear in use is staggering. For example, considering only gillnets in the North Pacific, at least 180,000 km of net is available to the various gillnet fisheries (Chen 1985; Gong 1985; Shima 1985; Uchida 1985), an amount that would stretch 4.5 times around the Earth. (This figure is conservative; it does not include gillnet in the coastal fisheries of Korea and Taiwan.) Even if only a small percentage, for example, 0.05% (Komatsu 1986), is lost in the course of these fishing operations, 90 km of gillnet would enter the North Pacific Ocean every time these nets are used. Gillnets may be lost as a result of storms, cut adrift by a ship crossing the float line, or discarded overboard after being damaged. Such derelict gillnets are a cause for concern because they may 1) continue to catch fish, leading to waste of marine resources and inaccurate estimates of fishing mortality; 2) present a hazard to navigation by fouling ships' propellers; and 3) ensnare and kill such nontarget species as seals, dolphins, whales, turtles, and seabirds.

The impact a piece of derelict gillnet will have depends on its size, shape, location, and length of time in the ocean. This paper reports the results of experiments designed to investigate some of these questions. Specifically, the objectives of the study were to measure the change in shape of derelict gillnet fragments of various sizes over time, to determine the fishing ability of derelict gillnets of known age, and to track the movement of drifting net fragments for periods up to 1 year.

METHODS

Thirty sections ("tans") of used, 113-mm monofilament gillnet of the type used in the Japanese high-seas squid fishery were purchased from Kyoei Unyu Company, Ltd., Hakodate, Japan. Each section measured 50 m long and 9 m deep, with floats at 1-m intervals. Sections were joined together to make four nets, 50, 100, 350, and 1,000 m long.

Attached to each of the four nets was a small, dual-frequency, radio-satellite transmitter buoy (Fig. 1), designed by Telonics of Mesa, Arizona. The buoy, 90 cm in length and 9 cm in diameter, allowed tracking and potential recovery of each net. The UHF satellite transmitter portion of each buoy used the Argos system (Argos 1984) to give a location on the Earth's surface accurate to within several hundred meters. The satellite transmitter broadcast on a schedule of 24 h on, 72 h off; a series of locations was, therefore, available once every 4 days. The VHF radio transmitter portion of each buoy allowed close-range directional tracking and recovery within a radius of approximately 10 km. The radio transmitter broadcast once a second without interruption. The combination of long- and short-range location systems was designed to allow physical recovery of the buoy and net after drifting freely in the ocean for up to 18 months.

To reduce windage and to avoid accidental discovery by fishermen or others, the buoy also was designed to be as inconspicuous, both visually and electronically, as possible. The buoy projected only 25 cm above the ocean's surface. Further, the megahertz frequencies transmitted by the buoy's location systems were beyond the kilohertz frequencies commonly used in ships' radio direction finders (RDF's) for locating buoys.

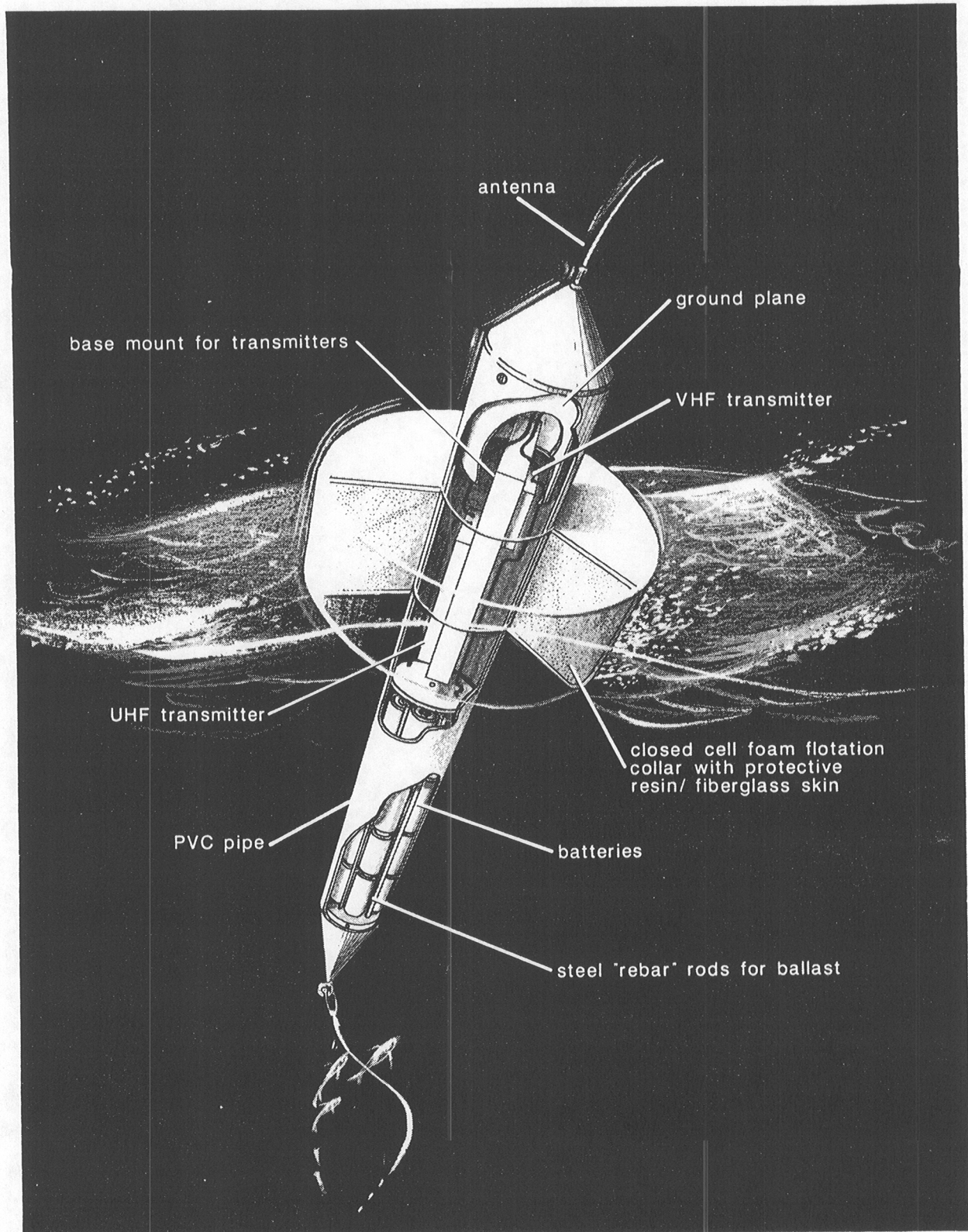


Figure 1.--Dual-frequency transmitter buoy used to track and recover experimental gillnets.

The nets and their associated buoys were deployed on 12 August 1986 from the NOAA ship *Townsend Cromwell* about 10 km east of Southeast Hancock Seamount, northwest of the Hawaiian Archipelago (Table 1). The nets were deployed by letting the ship drift downwind; hence, all nets were initially set parallel to the wind with the transmitter buoy at the downwind end. Measurements of net heading, length, and catch were made three times a day for 3 days from an inflatable boat. A temporary buoy was attached to one end of the net to serve as a visual target; from the other end, heading was then measured with a hand-held bearing compass, and length with an optical range finder. The configuration of each net was sketched. The longest (1,000-m) net was surveyed in a similar way, except that several visual targets were placed along its length, and measurements made in sections. Catch and fish aggregation around the nets were monitored by snorkel or scuba diving and documented photographically with video and 35-mm cameras.

Observations and measurements of the nets were confined to daylight hours. To track the nets during the night, larger buoy systems were attached to some nets each night and removed the next morning. Such a system consisted of a large RDF transmitter buoy; a long bamboo pole buoy with strobe light on top and large inflatable float; and a small, round, plastic buoy at the end of a tag line. The whole system had considerable windage. Because such a system probably affected a net's dynamics, the periods during which a large transmitter buoy system was attached to a net were considered in the interpretation of the results. The small radio-satellite transmitter buoys (Fig. 1), which were attached to the nets at all times, were considered to have negligible effects.

On the 10th day after deployment, the *Townsend Cromwell* returned and relocated all four nets. Rough seas, however, prevented launching a small boat, and the nets had to be observed from the deck of the *Townsend Cromwell*. After the cruise, the buoys were tracked by satellite until each buoy was either recovered or the signal from the satellite transmitter was lost. Positions were determined from monthly reports of Service Argos, Toulouse, France.

Table 1.--Experimental gillnet deployment on 12 August 1986 and tracking in the central Pacific Ocean. Dates and times are Midway standard time.

Buoy No.	Net length (m)	Deployment			Recovery			Days tracked (No.)
		Time (h)	Latitude N	Longitude E	Date	Latitude N	Longitude	
10013	50	0841	29°46.7'	179°10.3'	11/3/86	28°48.9'	176°57.6'W	83
10010 ^a	100	0945	29°46.8'	179°09.8'	1/7/87	28°14.2'	178°13.2'W	148
10011 ^a	350	1226	29°47.0'	179°08.6'	10/8/86	29°35.2'	176°18.9'W	57
10012	1,000	1330	29°47.6'	179°08.0'	6/17/87	23°00.1'	178°00.0'E	309

^aBuoy was not recovered; recovery data reflect time and location at which the signal was lost.

RESULTS

Shape and Heading

During the first 3 days, fair weather and calm seas greatly aided tracking and observation of the nets. Wind was east-southeast during this initial period but shifted to east-northeast on the second and third days and rose slightly in strength (Table 2). The nets first drifted north-northwest, then north-northeast, traveling about 9 km/day.

The 50- and 100-m nets shortened soon after deployment (Fig. 2). The 50-m net, in fact, had already collapsed by the time of the first observation, 30 min after deployment. "Collapsed" means that the net was folded like an accordion and all floats were close together. The net, however, was still hanging freely in the water; it was not tangled with itself.

The 350-m net contracted to about 40% of its original length during the first few hours, but then contracted more slowly (Fig. 2). By the 10th day, it had collapsed completely. The rate of collapse of this net may have been affected by the large transmitter buoy attached to the downwind end during the first night. The net was slightly longer the next morning (observation at 21 h). After removal of the large buoy system, the net further contracted (25 h) but was longer in the evening (29 h). The next day, the net followed a similar pattern, contracting between morning and afternoon and lengthening by evening. Greater detail of the changes in configuration of this net is shown in Figure 3A. Interestingly, the net rotated so that its heading 50 h after deployment was approximately 140° from its original heading.

Table 2.--Summary of wind and swell observations on 12-14 August 1986, the first 3 days of the gillnet experiment. Data are means calculated from the ship's hourly weather log.

Date	Time (h)	Wind		Swell	
		Speed (kn)	Heading	Height (ft)	Heading
12 August	0100-1200	9	116°	3	121°
	1300-2400	11	115°	3	116°
13 August	0100-1200	10	100°	3	118°
	1300-2400	14	70°	4	98°
14 August	0100-1200	13	74°	3	88°
	1300-2400	13	68°	4	83°

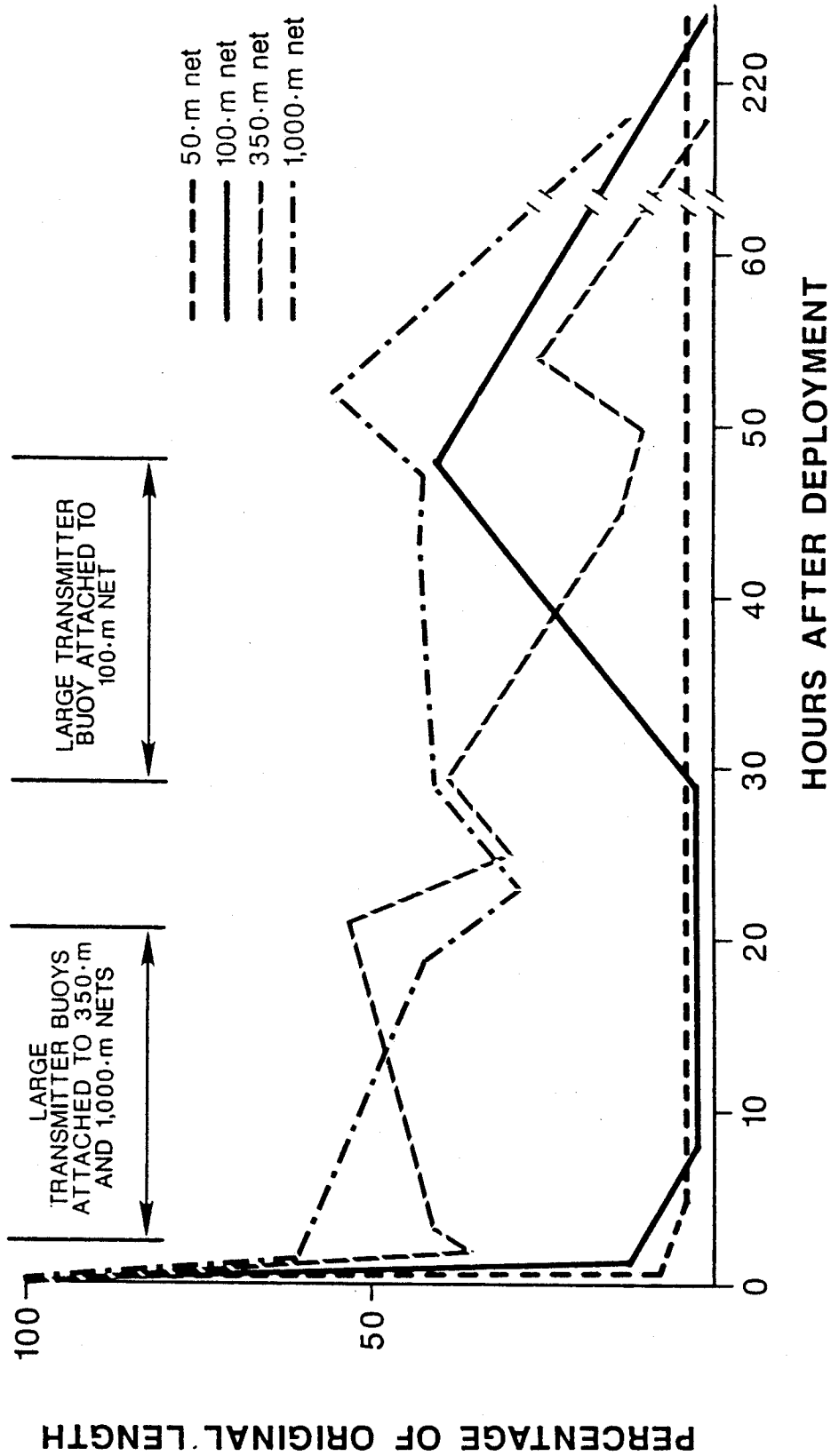


Figure 2.--Percentage of original lengths of four experimental derelict gillnets over time. During the first and second nights of observation, large transmitter buoys attached to certain nets may have affected net lengths.

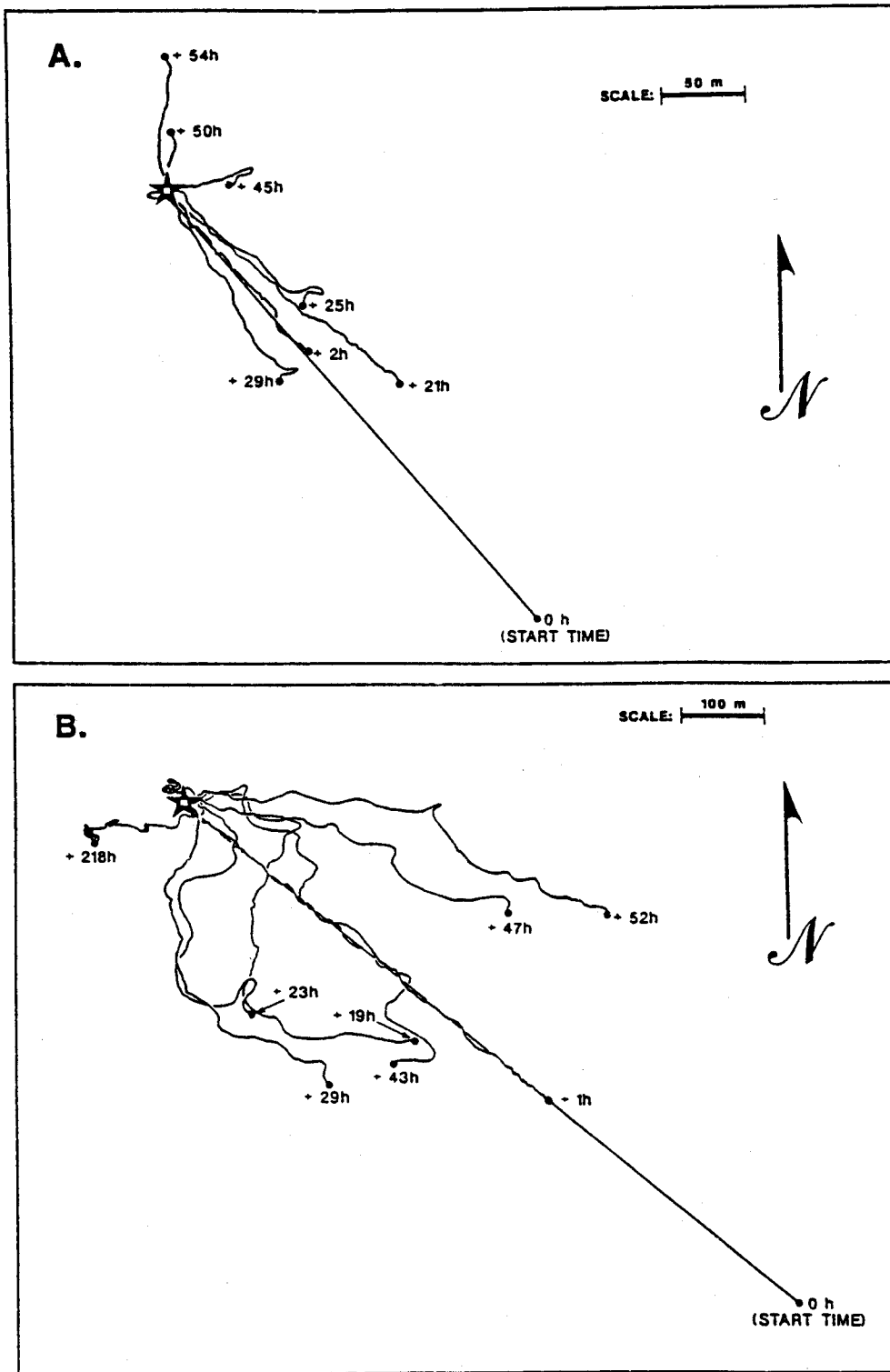


Figure 3.--Shape and heading of two experimental derelict gillnets at various points in time after deployment. The original windward end of the net (without the transmitter buoy) was placed at the origin (★) of each sketch. A) 350-m net. B) 1,000-m net.

The 1,000-m net followed a pattern of contraction and expansion similar to the 350-m net, except that its relative size 52 h after deployment was only about 50% of original length instead of 25% (Fig. 2). The 1,000-m net also rotated in the same direction as the 350-m net, but only by about 30° during the same period. By the 10th day (218 h), its heading had changed completely (Fig. 3B).

The time required for a gillnet to collapse was related to its original length. Figure 4 shows the time required for the various fragments of net to collapse to 10% of their original length as a function of their original length.

Catch

Very little was caught in any of the gillnets during the initial 3 days of observation. On the morning of the second day, a small marlin (*Makaira* sp.), about 1 m total length, was entangled in the 1,000-m net at the surface. On the third day, a large flyingfish (Exocetidae) was similarly caught in the same net. None of the other nets had any animals entangled in them by the end of the third day. Three small kahala (*Seriola* sp.) were observed swimming around the 350-m net on the second day. One opelu (*Decapterus* sp.) and three small kahala were seen around the 1,000-m net on the third day. Soon after the nets were deployed, several albatross (*Diomedea* sp.) landed on the water near the float line, but each left after a short investigation.

After 10 days, nothing was visible in the 50-, 100-, or 1,000-m nets, although one mahimahi, *Coryphaena hippurus*, was swimming near the latter. The floats of the 350-m net were in a tight group with numerous small kahala swimming nearby. A rotting, 2-m shark of undetermined species was entangled in the net, together with several bony fish too rotten to identify.

Movement

The location of each net during the entire course of the study, plotted once every 4 days, is shown in Figure 5. The number of days the buoys were tracked ranged from 57 to 309 (Table 1). The buoys and nets stayed in the general vicinity of the northwestern end of the Hawaiian Archipelago. For several months they remained north of Midway, then moved south. After 83 days at sea, the 50-m net and buoy 10013 were recovered by the *Townsend Cromwell*. Several species of fish were swimming near the net, and two pilotfish, *Naucrates ductor*, were caught in it (Table 3). No large animals were entangled in the net.

Buoy 10012, which was tracked the longest, traveled as far south as lat. 17°37.8'N, then returned north and west (Fig. 5). It was recovered after 309 days at sea by the chartered fishing vessel *Feresa*. The 1,000-m net was no longer attached to the buoy at that time. It is not known when the net became separated from the buoy, but the absence of barnacles, together with damage to the buoy, suggested that separation may have occurred only a short time before recovery. The buoy failed to transmit a

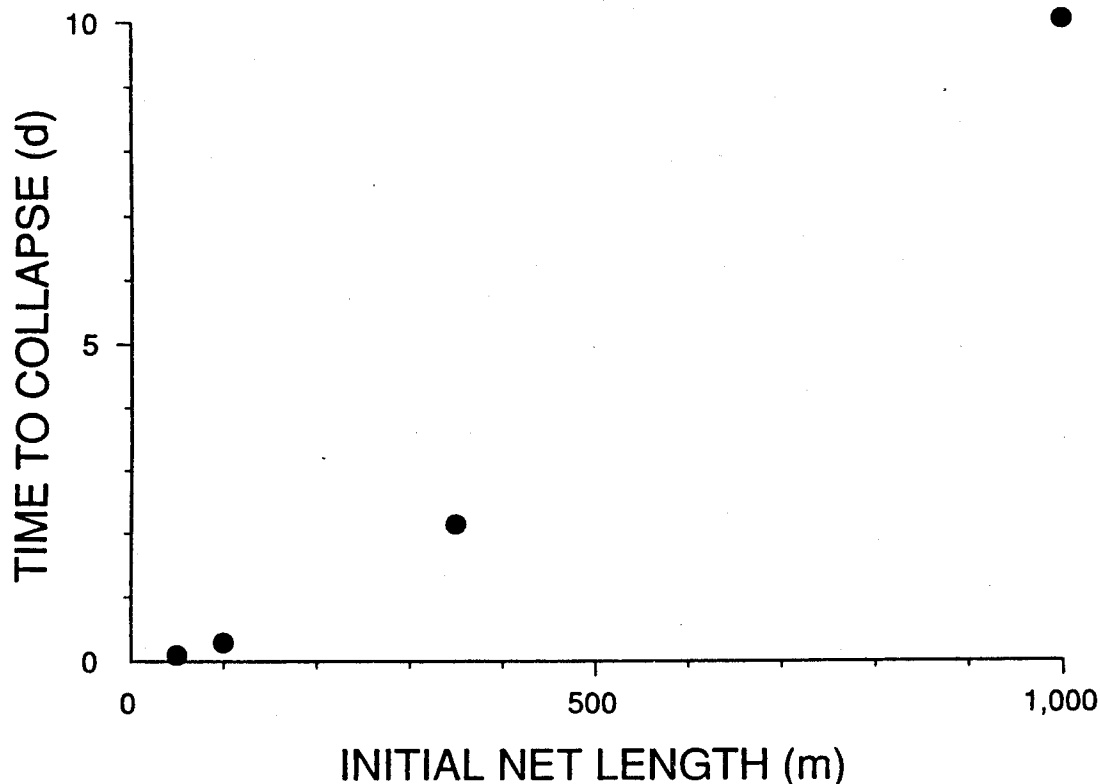


Figure 4.--Time required for a fragment of gillnet to collapse to 10% of original length as a function of its original length.

position on 8 June 1987, 9 days before recovery (Fig. 5), and was possibly entangled in the net at that time.

The two remaining nets and buoys were not recovered because their signals were lost. Signals stopped after 57 days for the 350-m net and 148 days for the 100-m net (Table 1). The reason for signal loss is not known, but the most likely explanation is that the buoys became entangled in the nets and submerged. Buoy 10010 on the 100-m net stopped transmitting about 15 km southeast of Kure Atoll (Fig. 5). Possibly the net became caught on the reef, but searches by plane and boat in August 1987 failed to find it.

For each 4-day interval, the mean speed of each buoy was computed. Mean speed per 4-day interval varied widely, from less than 1 km/day to nearly 50 km/day (Fig. 6). The 4-day mean speeds reflect several types of water movement: advection, inertial movement, and other eddies of various scales. Overall mean speed was 14.8 km/day or about 0.3 kn. The frequent and abrupt changes in speed and direction, however, meant that the distance

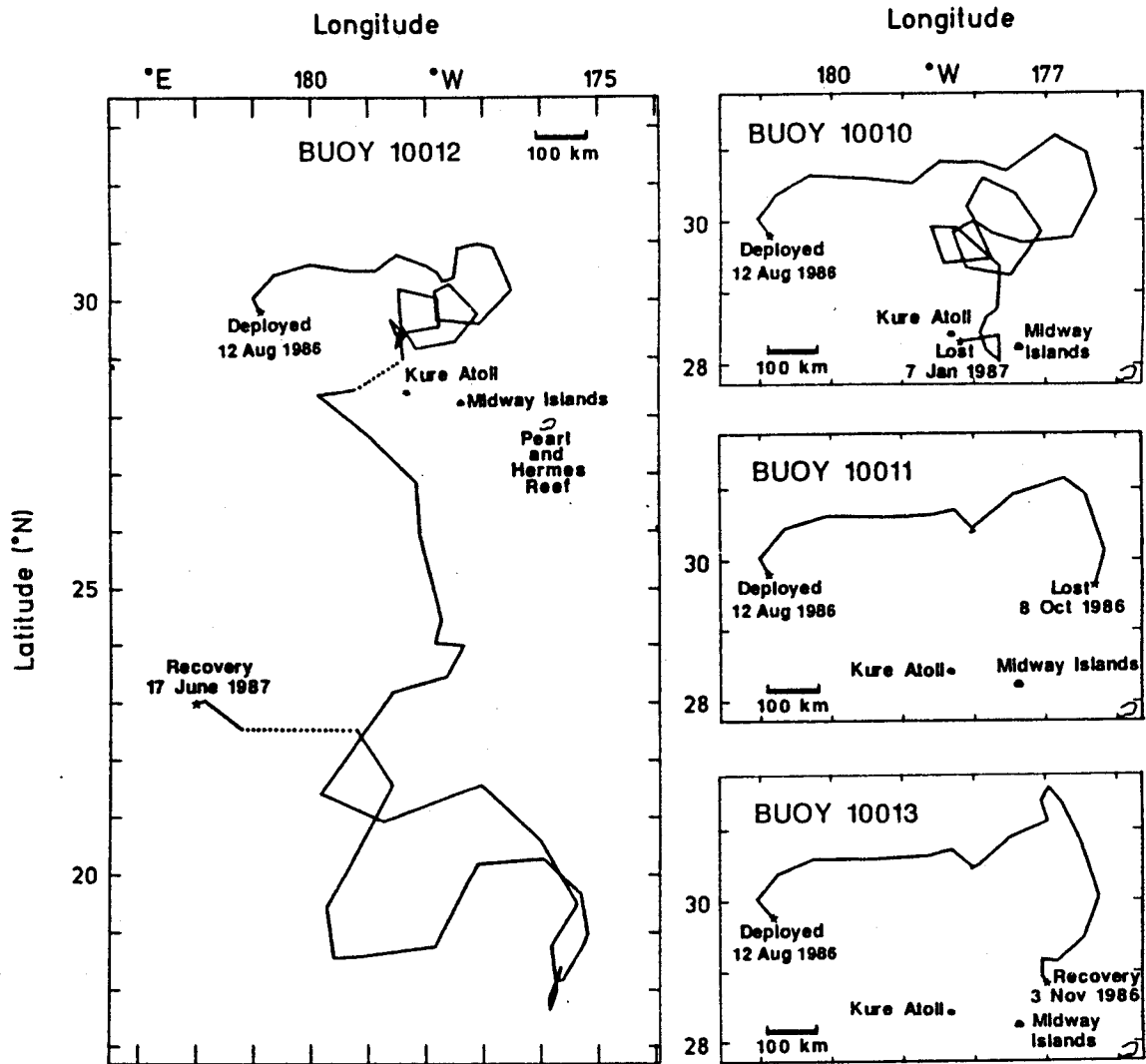


Figure 5.--Positions of four experimental derelict gillnets in the central Pacific Ocean, plotted at 4-day intervals. Dotted lines connect positions more than 4 days apart.

from the point of release did not bear any simple relation to time (Fig. 7). The four buoys traveled more or less together as long as they were tracked; the buoy tracked the longest (buoy 10012) drifted nearly 1,500 km from the point of deployment, then returned (Figs. 5, 7).

DISCUSSION

The amount of gillnet that becomes lost, detached, or discarded in the course of gillnet fishing operations is not known with any precision. Based on fishing activity, however, the total amount is undoubtedly large. The loss rate of 0.05%, mentioned earlier, is an unsubstantiated estimate given by a Japanese Government official (Komatsu 1986) during public hearings on the incidental catch of marine mammals during high-seas driftnet salmon fishing. Eisenbud (1985), citing a 1982 letter from Richard B. Roe,

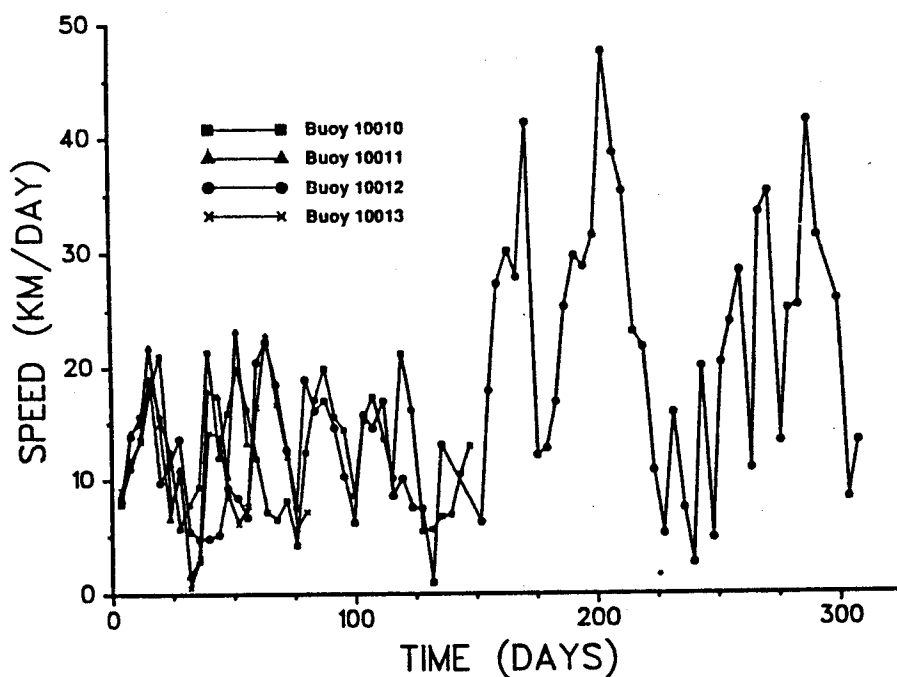


Figure 6.--Mean speed in each 4-day interval for the four gillnets, plotted as a function of time from deployment.

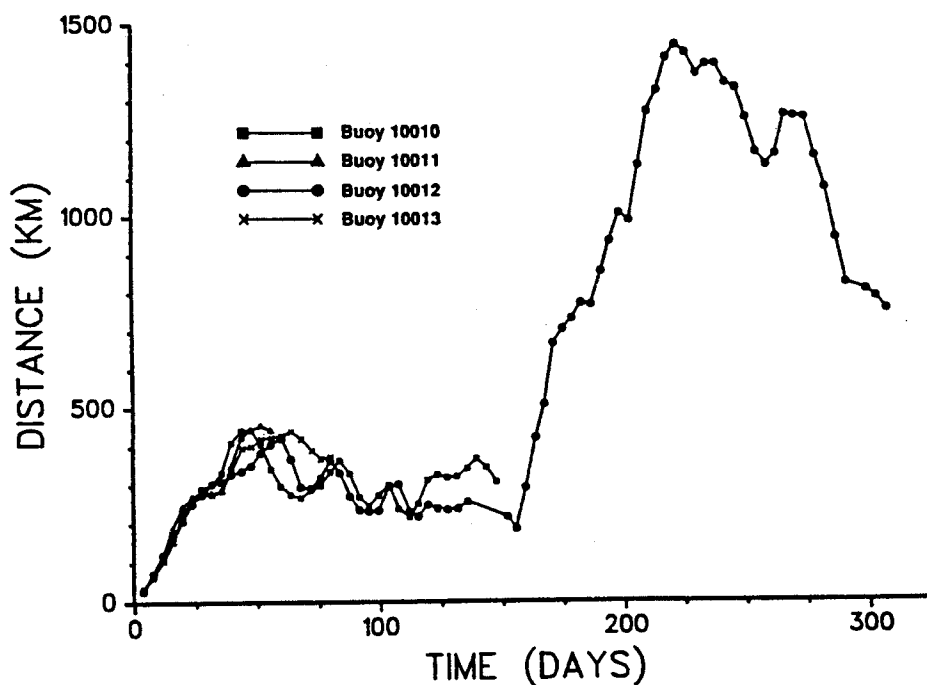


Figure 7.--Distance of each fragment of gillnet from the point of deployment, plotted as a function of time from deployment.

Table 3.--Biological observations near the 50-m gillnet recovered on 3 November 1986 north of Midway. The net had been drifting for 83 days. *N* = number of individuals sighted.

Species sighted	<i>N</i>	Sighting location
Pilotfish, <i>Naucrates ductor</i>	2	Caught in net.
Barracuda, <i>Sphyraena helleri</i>	20	Swimming near net.
Mahimahi, <i>Coryphaena hippurus</i>	2	Swimming near net.
<i>Alutera scripta</i>	11	Swimming near net.
<i>Naucrates ductor</i> juveniles	14	Swimming near net.
Unidentified fish, possibly <i>Kyphosus</i> sp.	7-12	Swimming near net.
Black-footed albatross, <i>Diomedea nigripes</i>	7	On surface near net.

NMFS, mentions an estimated 0.06% loss rate for the same fishery. Such a low loss rate, even if accurate for this fishery, is unlikely to apply to other types of gillnet fishing. For example, coastal gillnetting operations are likely to have a higher rate of net loss because of more boat traffic and a greater chance of nets becoming hung up on the bottom. Even so, applying this minimum loss rate to the total amount of available gillnet means that thousands of kilometers of derelict gillnet enter the North Pacific every year.

The fishing ability of a net depends on its size and configuration in the water (see also Mio et al. 1990). Left alone, a drifting gillnet will eventually collapse and become entangled with itself. The rate at which this happens depends, among other things, on the original length of the net. Rapid collapse of short sections of net is expected because of the weight of the lead line and local turbulence; longer nets have greater resistance to these small-scale effects. Over the range of sizes of gillnet fragments used in this study, it appeared that the rate of collapse was approximately 100 m/day (Fig. 4). Thus, net fragments less than 100 m long collapsed in less than a day, those several hundred meters long in several days, and those 1 km or longer in several weeks. These rates give a first approximation of the length of time a derelict gillnet would remain in an active fishing configuration. Note that these estimates apply to *intact* fragments of gillnet--that is, with both float and lead lines attached. The absence of a lead line, in particular, might affect the rate at which a net fragment collapses.

The rate of collapse may also depend on other factors. High wind and swell may make the net collapse faster. If a large animal, such as a shark or seal, is caught in a net, its struggling may also hasten the collapse of the net. If a buoy is attached to one end of the net, the force of wind on the buoy may keep the net open much longer, as demonstrated by the effect of the large transmitter buoy system on the 100-m net in this study. After 1 day, the net was completely collapsed, but after the large buoy system was attached to it overnight, the net lengthened (Fig. 2). The nearby 50-m net, which did not have a large buoy system attached to it, did not

lengthen during the same period. The force of the wind on the large buoy, which was at the downwind end of the net, caused a constant pull on one end of the net and was the likely cause of its lengthening.

Once collapsed, a gillnet is still capable of catching fish, though much less effectively. The rapid collapse of the nets in this study suggests that the catch rate of a lost or discarded gillnet will, for the target species, decline rapidly. Whether the hazard of a derelict gillnet also declines rapidly for nontarget species, however, is not resolved by this study. A floating mass of net will attract fish that may, in turn, attract predators like birds, sharks, seals, and dolphins.

The movement of debris on the ocean's surface is controlled by a combination of wind and surface currents. The gillnets used in this study have a large surface area in the water and little above it. Hence, their movement over a period of months (Fig. 5) reflects mainly the movement of the upper 10 m of water rather than wind drift. Currents in the Hawaiian Archipelago are complex and irregular (Wyrcki et al. 1969). Eddies of various sizes are common in Hawaiian waters (Seckel 1955; Patzert 1969), and the loops executed by buoys 10010 and 10012 may indicate such eddies. Movements on smaller space and time scales, such as inertial motion, are not resolved by the 4-day interval between buoy positions in this study. Inertial circling was observed in the finer scale measurements of Matsumura et al. (1990).

The abrupt changes in speed and direction of the nets in this study illustrate that predicting the movement of marine debris is a difficult problem (Galt 1985; Seckel 1985). The movement of marine debris can be approached both experimentally and through simulation modeling (Matsumura et al. 1990). At least around the Hawaiian Archipelago, simple models of linear motion (distance proportional to time) or diffusion (distance proportional to the square root of time) will not predict the movement of derelict gill nets (Fig. 7). The general problem of predicting the movement and fate of debris in the ocean requires greater knowledge of factors affecting the "birth" and "death" rates of the various "species" of marine debris (Gerrodette 1985).

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PRELIMINARY STUDY ON CHANGE IN SHAPE OF DRIFTING
NETS EXPERIMENTALLY PLACED IN THE SEA

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ABSTRACT

Research activities to understand the impact of lost drifting nets on marine organisms were initiated in 1988, and an experiment to clarify how the lost nets change their shape at sea was conducted as the first stage of the activities.

Five driftnet sets (40 tans each) were placed in the water in the area around lat. 38°N, long. 158°E, and their shapes were observed from 5 to 25 May. After this, the net sets were allowed to drift, and about 3 months later (in early September) were again observed and subsequently retrieved.

The observations were visual, and recordings were made using a camera-equipped balloon and a video camera from a research vessel.

Three days after setting, one of the nets began twisting into a mass near each end of the net. As time passed, the mass grew larger: the ends of the net approached each other and the net folded in half. Each mass continued to grow, and several small masses also were formed in portions of the long, overlapped net. Twenty days after setting, the net had become one large mass.

All nets observed became masses in the same way, although the speed of formation varied. In September, when the research vessel visited these nets again, each was found floating in a mass.

INTRODUCTION

It has been noted that fishing nets, especially gillnet fragments (hereafter referred to as drifting nets), drift in the sea out of man's control and continue catching marine organisms such as fish while drifting. However, there has been only fragmentary information concerning movements of drifting nets and the actual damage done to marine organisms. Therefore,

since 1988 we have been conducting research on the movement and changes in shape of drifting nets as well as their impact on marine organisms. This paper examines the changes in the shape of drifting nets in the course of time. Gillnets are thought to be the most effective means of catching fish, as they are set in the sea in a straight line. Once freed from man's control, gillnets are believed to change shape as time passes, with their fishing efficiency gradually declining. We conducted a survey on changes in the shape of drifting nets over time in order to establish a basis for research on related changes in fishing efficiency.

METHODS OF SURVEY

The research was conducted in the North Pacific Ocean on the salmon driftnet fishing ground at lat. 35°-45°N, long. 150°E-180°. The first survey was conducted from 1 to 30 May 1988, and the second from 17 August to 30 September. Five nets were used for the experiments, and each consisted of 40 tans of nylon monofilament gillnet with a mesh size of 115 mm.

An Argos buoy and a "self-call" buoy were attached to the net, one at each end. The location of each net was recorded an average of six times a day using information from the Argos buoy.

On the first cruise, experimental drifting nets were observed by sighting from on board the vessel and photographing from above with a remote control camera attached to a balloon. On the second cruise, experimental drifting nets were located using information from the buoy. They were retrieved after visual confirmation as well as confirmation through a remote control television attached to a balloon.

RESULTS

The six experimental nets, stretched tight, were set in the area lat. 39°20'-38°43'N, long. 154°33'-155°44'E from 1406 on 6 May to 1710 on 7 May. The experimental nets were observed a total of only 16 times, since they moved in two different directions after setting, with two drifting northeast and the other three southeast. Three nets were observed four times and two were observed twice before being retrieved (Table 1).

Except for net No. 1, each experimental net showed generally the same pattern of changing although they differed in pace. First, each end of the net twisted and formed a small mass (Fig. 1A). Second, each net folded in half and its two ends approached each other. The two ends formed a mass, twisting with each other, and the rest of the net stretched long, overlapping more and more (Fig. 1B and C). Third, as time passed, the stretched part wound around the mass. After reaching the third stage, the stretched part of the net formed a mass slowly, becoming entangled and disentangled. Observed 15 and 18 days after release, it was 50 to 60 m long compared with its original length of 2 km, indicating that it did not need many days to become a complete mass. When the five experimental nets were all collected after drifting for a long time, each net had formed a complete mass (Fig. 1D).

Table 1.--Trajectory and width ($\leftarrow \rightarrow$) change of six floating nets in 1988.

Net No.	Date	Width (m)	Latitude N	Longitude E
1	25 May	2,000	40°04'	153°06'
	15 Sept.	5	40°20'	161°08'
2	7 May	2,000	39°19'	154°50'
	10 May	1,250	39°10'	155°12'
	12 May	120	39°19'	155°25'
	18 May	250	38°12'	159°04'
	23 May	60	38°31'	158°15'
	3 Sept.	5	33°56'	169°15'
3	7 May	2,000	39°05'	155°28'
	11 May	160	38°14'	156°52'
	19 May	130	37°18'	160°14'
	28 Aug.	5	40°51'	171°28'
4	7 May	2,000	38°42'	155°19'
	12 May	310	38°14'	154°55'
	18 May	120	37°29'	158°49'
	31 Aug.	5	35°34'	179°46'
5	7 May	2,000	38°48'	154°04'
	13 May	800	39°25'	153°58'
	16 May	500	39°23'	153°56'
	21 May	250	40°06'	153°45'
	25 May	50	40°12'	153°05'
	11 Sept.	5	39°57'	158°10'
6	6 May	2,000	39°06'	154°33'
	8 May	1,150	39°20'	154°28'
	10 May	1,080	39°34'	153°23'
	16 May	600	39°19'	158°07'
	22 May	180	39°32'	153°19'

Note: Measurement of width ($\leftarrow \rightarrow$) refers to Figure 1.

As for the time required to reach each stage, the five nets (excluding net No. 1) remaining in this experiment can be divided into two groups (Fig. 2): One group needed 4 to 5 days after release to reach the third stage; the other needed 14 to 16 days to reach the third stage.

The approach of both ends of a drifting net is the basic process of changing the shape. The structure and arrangement of nets and accompanying buoys also seem to affect the changes. In this experiment, buoys were

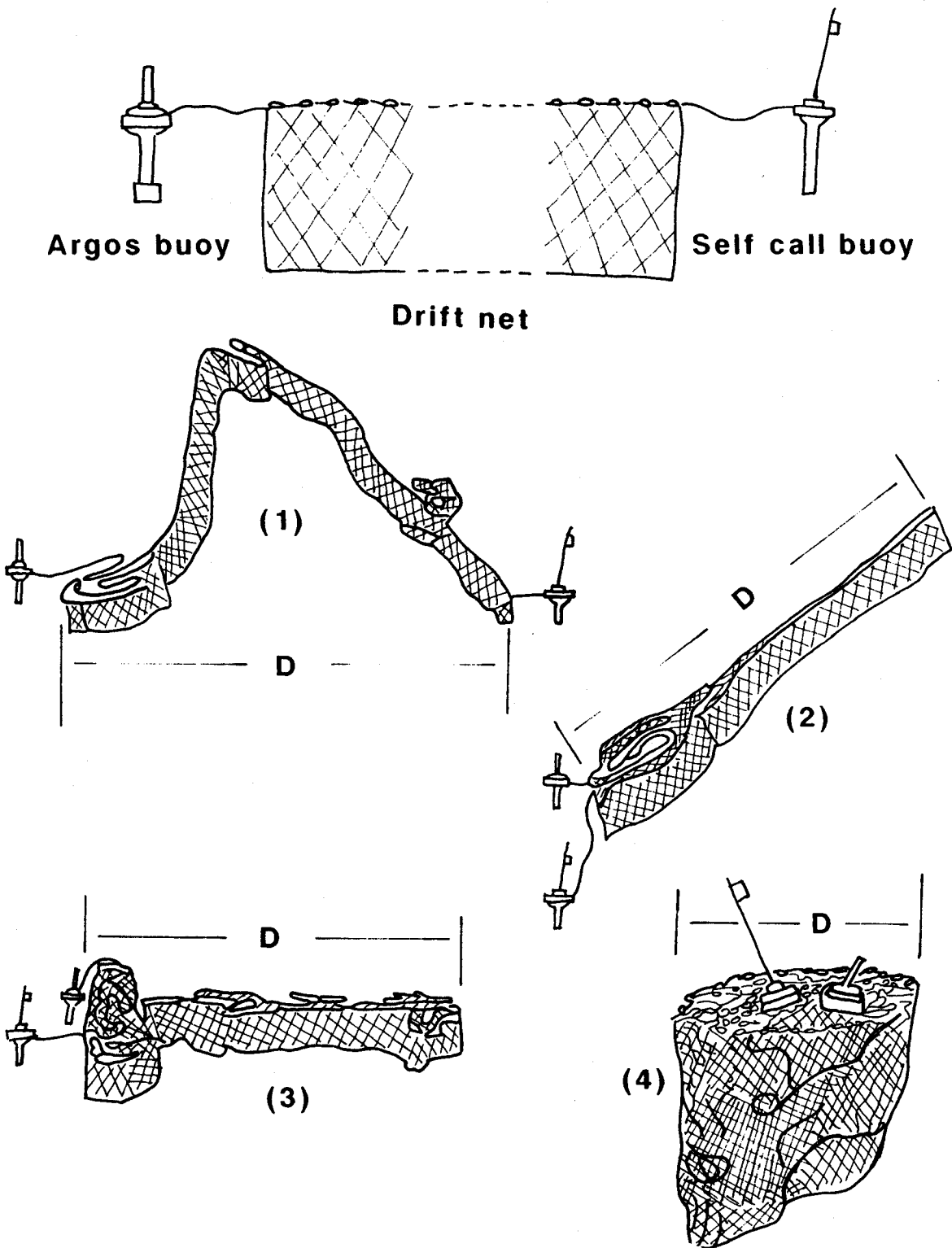


Figure 1.--Schematic diagram showing formation of a mass of floating net after setting ((← →) denotes width of floating net).

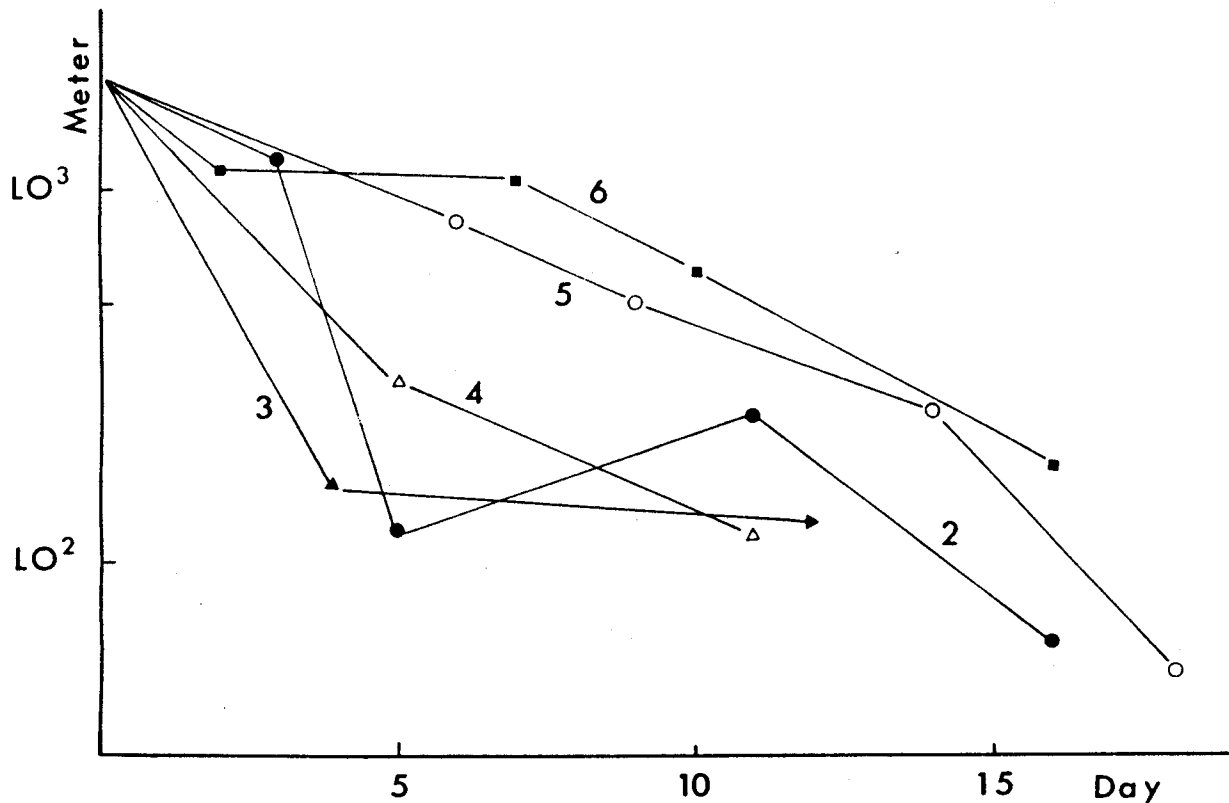


Figure 2.--Relationship between floating period and longest width of net.

attached at both ends of a net. It may be assumed that the difference in resistance between the buoys and the net helped the two ends approach each other. The Argos buoy is a cylinder 160 mm in diameter and 790 mm in height, with a float 440 mm in diameter and 200 mm in height. It is quite small compared to the size of the net. Eight days into the experiment, the Argos buoy temporarily attached to No. 4 net dropped off, and one end of the net was subject to the same resistance as the net alone. However, both ends of the No. 4 net approached each other in the same way as the other nets.

As both ends behaved in a manner similar to other nets with buoys attached on both ends, it is suggested that there would be no changes in the basic configuration even though the presence of buoys may affect the speed of shape change. We plan to conduct further experiments to study effects of different conditions on net shape changes.

